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MOUNTAIN SHEEP AND MINING: IMPLICATIONS FOR CONSERVATION AND MANAGEMENT

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Opportunities to quantitatively assess responses of ungulates to mineral extraction have been limited. Reasons for this dearth of research include a lack of adequate funding, available personnel, and logistical constraints. In 1992, a request was submitted to the Bureau of Land Management by a mining company for permission to extract and process gold ore in the Panamint Range, Inyo County, California, near a spring presumed to be critically important to mountain sheep, *Ovis canadensis*. Ensuing compliance with the National Environmental Policy Act resulted in funds to monitor effects of mining activities on mountain sheep inhabiting that area. Because funding was not released until ~8 months prior to construction and operation of the mine, we were unable to adequately address the pre-mining ecology of sheep in the "affected" area. We therefore employed a simultaneous treatment-control study designed to test several hypotheses regarding effects of mining activities on habitat selection, demographics, home-range dynamics, foraging activities, and composition and quality of diet for mountain sheep during 1995-1997. During our 3-yr study, we radiocollared and monitored 86% (n = 19) of all adult female sheep known to exist within the mined (treatment) and nonmined (control) areas. Size of annual home ranges, composition of diet, and ratios of young to adult females did not differ between female sheep

inhabiting mined and nonmined areas. The nonmined area contained more annual plants, succulents, and perennial forbs than did the mined area, whereas abundance of shrubs, quality of forage, and relative abundance of carnivores did not differ between sites. During spring, female sheep adjacent to the mine spent more time foraging and had a lower-quality diet than those in the nonmined area. Conversely, during summer and autumn, female sheep from the mined area spent less time foraging than those in the nonmined area, but continued to have a lower-quality diet. All females were nearest water in summer compared with other seasons. During all seasons, females selected sites with more mixed-woody scrub, lower elevations, steeper slopes, and less visibility than available at random locations. We observed the greatest disparities between study areas in time spent foraging and diet quality during summer. In summer, females from the mined area were nearest to the mine; amount of explosives used, frequency of blasting, and amount of ore hauled from the mine were greatest during that period. Because of their reliance on a source of permanent water adjacent to the mine during summer and autumn, we hypothesize that female sheep from the mined area spent more time vigilant during those seasons and, consequently, less time foraging than conspecifics in the nonmined area. If outcomes we observed persist for mountain sheep in the mined area, reduced nutrient intake could have demographic consequences for that subpopulation. Thus, providing a reliable source of water away from the mine, or reducing mining activity during summer, may benefit mountain sheep that currently use areas adjacent to the mine.

INTRODUCTION

Among ungulates, exposure to human activities has been linked to temporary abandonment of areas of traditional use (Kuck et al. 1985, Bleich et al. 1994), shifts in centers of activity (Van Dyke and Klein 1996), and localized extirpations (DeForge et al. 1981). Quantitative data regarding effects of mineral extraction on mountain sheep, *Ovis canadensis*, however, are few. Some researchers have investigated responses of mountain sheep to activities typically associated with mining (e.g., helicopter disturbance—Stockwell et al. 1991, Bleich et al. 1994; human disturbance—Hicks and Elder 1979, Leslie and Douglas 1980, Krausman and Etchberger 1995, Papouchis et al. 2001; and water development—Krausman and Etchberger 1995), but those authors did not address the issue of mining activities and their effects on mountain sheep.

We studied the ecology of female mountain sheep adjacent to a heap-leach gold mine in the Mojave Desert. We measured variables from two subpopulations of female sheep inhabiting distinct geographic areas within a single mountain range (i.e., mined and nonmined) to test hypotheses regarding potential influences of mining on habitat selection, home-range dynamics, and foraging ecology of those large herbivores. We predicted that if mining had no effect on sheep, there would be no differences in selection of habitat and size of home ranges between females occupying those two areas. We also predicted that after considering potential influences of resources and

predators, female sheep would select habitat in a manner similar to that of sheep from the nonmined area, if there were no effects of mining activities. Among mountain sheep, vigilant behavior increases in areas with low levels of visibility (Risenhoover and Bailey 1985, Frid 1997, Rachlow and Bowyer 1998), and is affected by the presence of perceived threats (Berger 1978, Festa-Bianchet 1988, Berger 1991, Stockwell et al. 1991)—such behavior reduces time spent foraging, and might result in differences between mined and nonmined areas. Therefore, if mountain sheep near the mine were unaffected by mining activities, we hypothesized that after considering availability of forage, there would be no difference in time spent foraging between areas. Further, if quality of forage differed between sites, we predicted that individuals consuming lower-quality forage would spend more time foraging to meet their nutrient requirements (Leslie and Douglas 1979). Finally, if abundance of predators differed between areas, we predicted that after considering availability and quality of forage, mountain sheep in areas with more predators would spend more time vigilant (Berger 1978, Rachlow and Bowyer, 1998) and, consequently, less time foraging (Molvar and Bowyer 1994, Bowyer et al. 2001). Effects of mining cannot be addressed or mitigated without considering influences of these factors on the ecology of free-ranging mountain sheep.

METHODS

Study Area

Our study was conducted in the Mojave Desert on the west-facing slope of the Panamint Range, Inyo County, California, USA (Fig. 1). The subpopulation of female sheep living near the mine was centered on Redlands Spring (36°56'37"N, 117°10'43"W) in the southern end of our study area, whereas the center of our control population (i.e., nonmined) was located ~22 km to the north (37°09'34"N, 117°09'50"W) (Fig. 2).

Elevations range from 305 m on the valley floor to 3,368 m at Telescope Peak. Mean ($\pm$ *SE*) annual rainfall from 1911 to 1994 at the weather station ~30 km from our study site (Greenland Ranch-Furnace Creek, California) was 4.7 $\pm$ 0.33 cm, and temperature was highly variable; daytime high temperatures ranged from >40°C during summer (May-August), to -7°C during spring (January-April, Fig. 3; Death Valley National Park Service files). We used climatological data collected from Greenland Ranch-Furnace Creek, and data on timing of parturition (Welles and Welles 1961) to define three seasons. Spring was 1 January-30 April, which incorporated most of parturition, and was typified by cool temperatures ($\bar{x}$ = 25° $\pm$ 5.7°C) and relatively greater precipitation ($\bar{x}$ = 0.59 $\pm$ 0.99 cm) than other seasons. Summer was 1 May-31 August; that period was extremely hot ($\bar{x}$ = 43° $\pm$ 3.8°C) with low rainfall ($\bar{x}$ = 0.21 $\pm$ 0.51 cm). Autumn (and the concomitant mating season) extended from 1 September to 31 December, and was characterized by cooler temperatures ($\bar{x}$ = 29° $\pm$ 9.9°C), and lower precipitation than spring ($\bar{x}$ = 0.37 $\pm$ 0.77 cm).

Six vegetation communities were delineated within the study area from a LANDSAT-TM scene with cells of 25-m resolution: 1) alkali playa (305 m elevation), which occurred on relatively flat areas of the valley floor; 2) desert saltbush, *Atriplex canescens*, scrub,

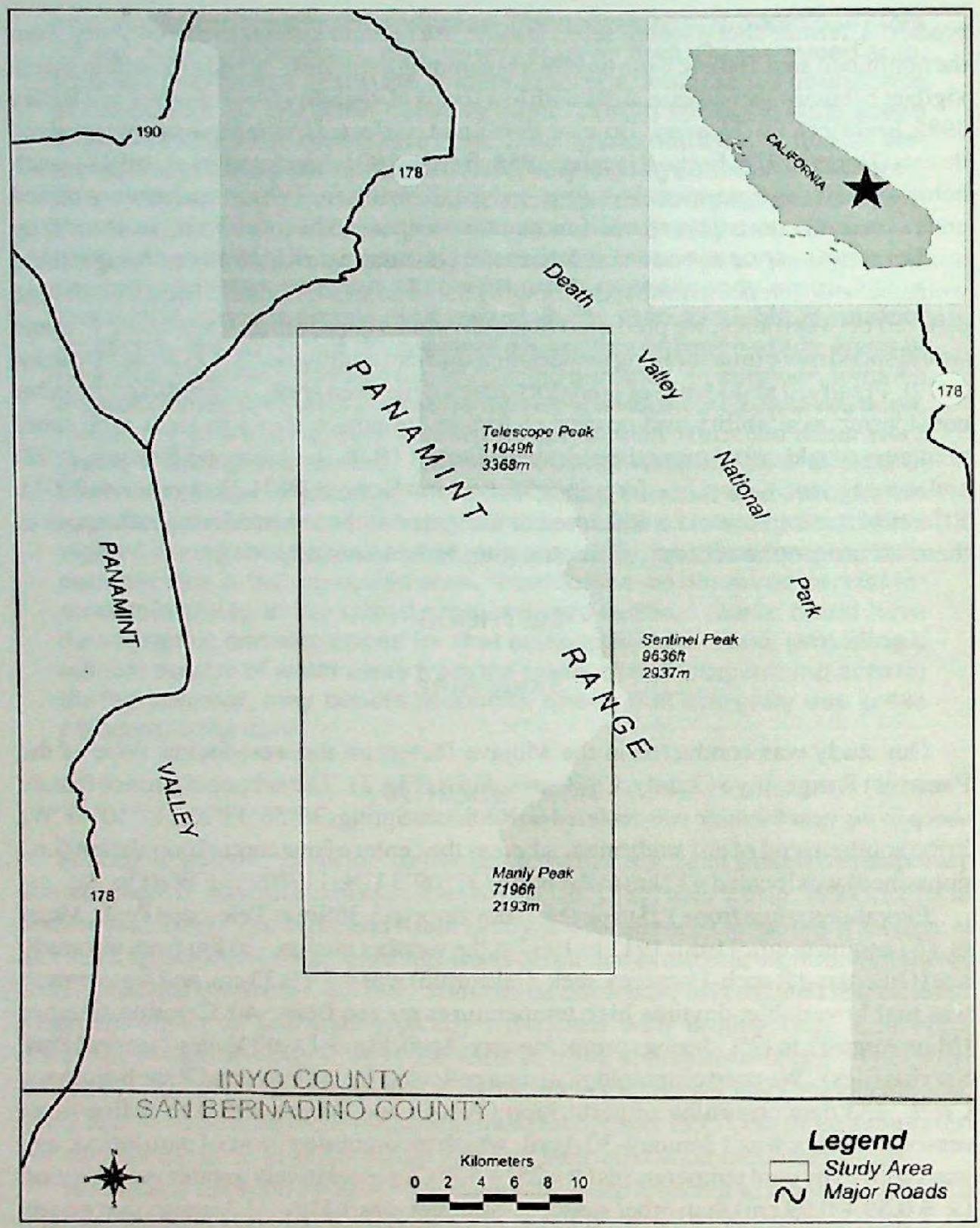

Fig. 1. Location of our study areas in the Panamint Range, Inyo County, California, USA, 1995-1997.

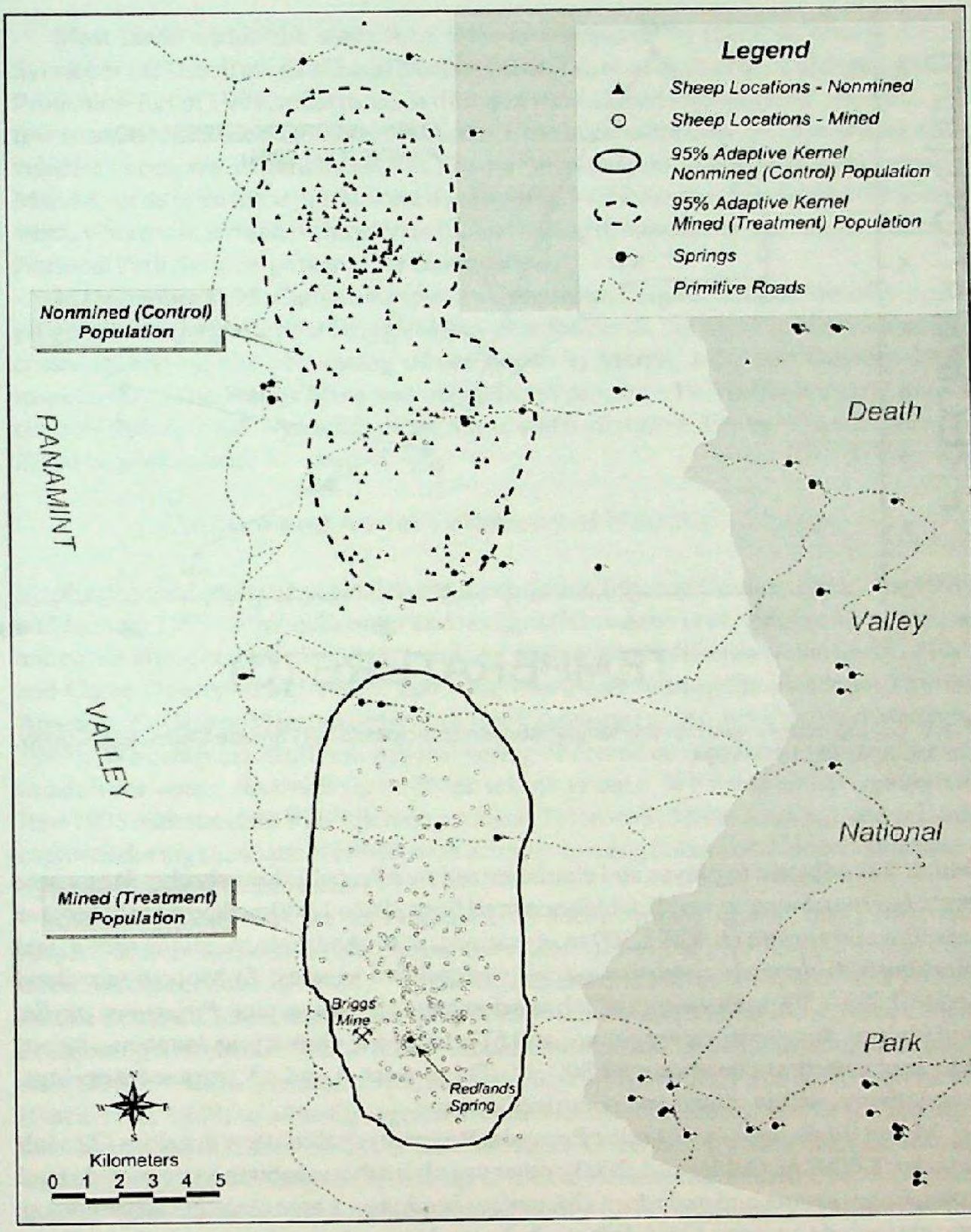

Fig. 2. Telemetry locations of female mountain sheep, and 95% adaptive kernel polygons for populations of sheep from mined and nonmined areas in the Panamint Range, Inyo County, California, USA, 1995-1997.

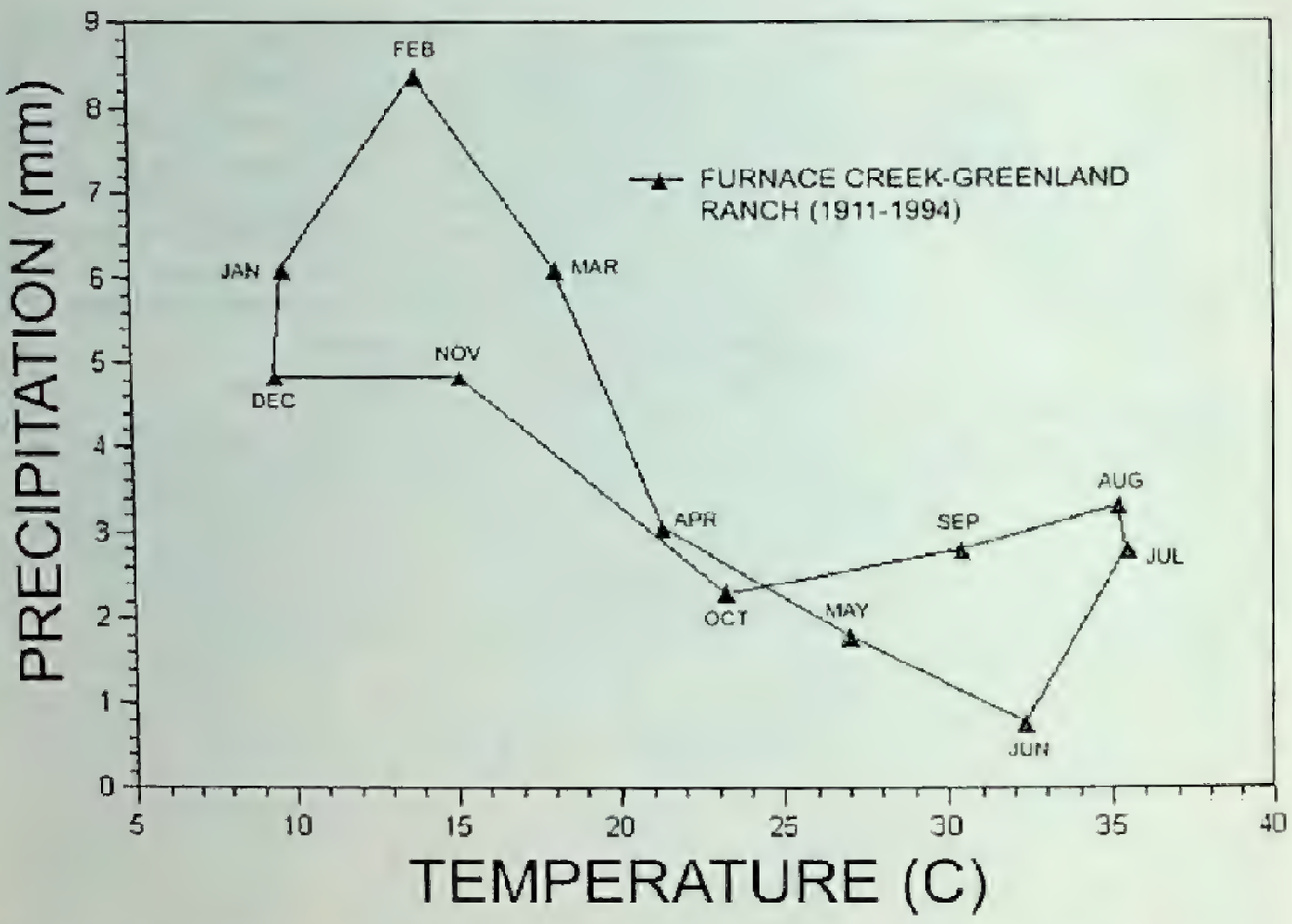

Fig. 3. Climograph of mean monthly temperature and precipitation at Furnace Creek, Inyo County, California, USA, 1911-1994.

which was adjacent to playas and characterized by microphyllous shrubs; 3) creosote bush, *Larrea tridentata*, scrub, which occurred from 300 to 1,640 m elevation; 4) Mojave mixed woody-scrub (1,400-2,300 m elevation), with shadscale, *A. confertifolia*, and blackbush, *Coleogyne ramosissima*, as predominant species; 5) Mojave-woodland scrub (2,300-3,300 m elevation), which was dominated by pinyon pine, *Pinus monophylla*, and juniper, *Juniperus osteosperma*; and 6) bristlecone pine, *Pinus longaeva*, forest, which occurred at elevations >3,300 m. There were 9 and 13 permanent springs, respectively, within mined and nonmined areas.

Mountain sheep occurred in the Panamint Range at relatively low densities (72 adult females/1,000 km; Ochler et al. 2003); other ungulates that inhabit the area include feral asses, *Equus asinus*, and mule deer, *Odocoileus hemionus*. Large mammalian carnivores present include coyotes, *Canis latrans*, bobcats, *Lynx rufus*, and mountain lions, *Puma concolor* (Welles and Welles 1961). Nevertheless, predation on desert sheep by those carnivores was thought to be negligible (Welles and Welles 1961, Weaver[1] 1972).

[1]Weaver, R. A. 1972. Desert bighorn sheep in Death Valley National Monument and adjacent areas. California Department of Fish and Game Wildlife Management Administrative Report 72-4.

Most lands within the study area were administered by the U.S. National Park Service or the U.S. Bureau of Land Management and, as a result of the California Desert Protection Act of 1994, most roads in this area were closed to motorized vehicles. The few roads not included in wilderness areas were accessible only by four-wheel drive vehicles; access was extremely limited. During the cooler portions of the year (November-March), roads open to the public were used by 10-15 vehicles/weekend, and 5-10 hikers/week, whereas in summer vehicular traffic and hiking in those areas was rare (D. Brenner, National Park Service, personal communication).

In December 1995, Canyon Resources Corporation began construction of an open-pit gold mine (hereafter the Briggs Mine) near Redlands Spring (Fig. 2); excavation, crushing, and on-site processing of ore began in March, July, and October 1996, respectively. The Briggs Mine was projected to process ~19.3 million metric tons of ore on site during the 7-year life of the mine, and to disturb 1,333 ha of land within the 2,350-ha project area.

Capture and Aerial Telemetry of Mountain Sheep

We captured adult (>1 yr old) female sheep during June and October 1995, June 1996, and January 1997 with a helicopter and net-gun (Krausman et al. 1985a); all aspects of animal handling complied with protocols set forth by the California Department of Fish and Game (Jessup et al.[2] 1986), and were consistent with methods adopted by the American Society of Mammalogists (Ad Hoc Committee on Acceptable Field Methods. 1987). We categorized all females and young observed or captured during fieldwork as adults or young (individuals of either sex <1 yr old). We fitted sheep captured in June 1995 with standard VHF telemetry collars (Telonics®, Mesa, Arizona), and animals captured during subsequent efforts with activity-sensing collars (Advanced Telemetry Systems®, Isanti, Minnesota).

We attempted to locate all radiocollared sheep weekly during June, July, and August, and in alternate weeks during the remainder of the year using a fixed-wing aircraft (Krausman et al.[3] 1984). We located collared sheep between 0900 and 1400 h, Pacific Standard Time, and locations were estimated with either LORAN-C or Global Positioning System (GPS) instruments aboard the aircraft. Because of error associated with LORAN-C (Jaeger et al. 1993, Oehler et al. 1996), we derived a correction factor (Patric et al. 1988) to adjust geographic coordinates obtained using that technology. On average, coordinates obtained from LORAN-C technology indicated the aircraft was 41 m west and 127 m north of the target. Global Positioning Systems are less subject to geographic variability in accuracy than LORAN-C (Leptich et al. 1994), and we did

[2]Jessup, D. A., W. E. Clark, and M. A. Fowler. 1986. Wildlife restraint handbook. Third edition. California Department of Fish and Game, Rancho Cordova, California, USA.

[3]Krausman, P. R., J. J. Hervert, and L. L. Ordway. 1984. Radio tracking desert mule deer and bighorn sheep with light aircraft. Pages 115-118 in Deer in the southwest: a workshop (P. R. Krausman and N. Smith, editors.). School of Renewable Natural Resources, University of Arizona, Tucson, USA.

not correct sheep locations obtained via GPS. Accuracy of telemetry locations was 177 m (i.e., a circle with radius of 177 m) for an investigation of mule deer in the San Bernardino Mountains, California (Nicholson et al. 1997). Because we used the same pilot as Nicholson et al. (1997) for our flights, we reasoned that the error within our study area would be similar. To be conservative, and because we occasionally used another pilot, we increased the radius of the circle to 200 m.

Habitat Analyses

Unless specifically indicated, hereafter our references to "habitat" are general in nature and apply to the suite of variables we analyzed relative to locations of sheep (e.g., slope, aspect, viewshed, vegetation communities, etc.). We tested locations for each animal for lack of independence with the multiresponse-sequence procedure (MRSP) of BLOSSOM statistical software (Solow 1989, Slauson et al.[4] 1991) and eliminated locations until no significant ($P \geq 0.05$) autocorrelation was detected (Nicholson et al. 1997). We pooled telemetry locations of female sheep by area (mined vs. nonmined) to define areas of available habitat and used the program CALHOME (Kie et al. 1996) to construct a 100% minimum convex polygon around those locations. Resulting polygons were buffered by 1,000 m to account for undetected movements (Bleich et al. 1997), and to avoid biases in assessing habitat selection from only within the home ranges of sheep (Kie et al. 2002). We then generated random locations within each buffered polygon with the same frequency as sheep locations used to construct that polygon. Next, we used a Geographic Information System (GIS; ARC/INFO®, Environmental Systems Research Institute, Redlands, California) to generate a circle with a radius of 200 m around each sheep (i.e., potential telemetry error) and random location; area within those circles was used to calculate relative use and availability of habitat attributes (Andrew et al. 1999, Nicholson et al. 1997).

We generated a three-dimensional model of terrain for the study area from USGS 7.5' digital elevation models (DEM) with 30-m resolution using the GRID module of ARC/INFO. The DEM provided information on elevation, slope, and aspect associated with each telemetry location. Because resolution of the DEM was 30 m, the radius of the circle used to assess associated features was 210 m (i.e., 30 m x 7 pixels). We used the product of the *SD* of slope and the mean angular deviation of aspect inside each circle as an index to terrain diversity (Nicholson et al. 1997). We used the GIS to estimate visibility (i.e., the viewshed) from each female sheep and random location to examine the role of visibility in habitat selection among female sheep. The GIS calculated the two-dimensional area that would be visible from a height of 1 m (approximate eye-level of a sheep) within a circle having a radius of 1,000 m.

A GIS layer of vegetation communities was developed for the study area from the LANDSAT-TM scene. Because alkali playa, desert-saltbush scrub, and the bristlecone-

[4] Slauson, W. L., B. S. Cade, and J. D. Richards. 1991. User's manual for BLOSSOM statistical software. United States Fish and Wildlife Service, National Ecology Research Center, Fort Collins, Colorado, USA.

pine forest represented <0.5% of the available plant communities, and because female sheep were not located in those vegetation types, we eliminated them from further consideration. We also digitized the locations of roads and springs from USGS 7.5' quadrangle maps to evaluate those factors relative to use of habitat by sheep.

We used stepwise logistic regression (PROC LOGISTIC; SAS Institute Inc. 1997) with an α-to-enter and stay of 0.15 (Agresti 1990) to identify variables that might be important in differentiating random locations from those used by sheep (Nicholson et al. 1997). For our habitat analyses, we treated the individual animals as our sampling unit to eliminate inflated sample sizes associated with using individual telemetry locations as sampling units. Using variables selected by logistic regression, we calculated a vector of means for each female sheep during each season (i.e., three vectors for each female). Because availability of the habitat variables we measured does not change by season, we calculated a single vector of means for those variables at the random locations in each of the mined and nonmined areas. Finally, we generated a vector of differences for each female sheep during each season (i.e., sheep minus the random vector from its corresponding area); resulting vectors became dependent variables in a two-way multivariate analysis of variance (MANOVA; Johnson and Wichern 1988) to test hypotheses regarding selection of habitat. Main effects in the habitat model were area (mined and nonmined) and season (spring, summer, and autumn); significance of the model was determined with Wilks' lambda (Johnson and Wichern 1988). We determined selection or avoidance of habitat variables following the methods of Nicholson et al. (1997) and Bowyer et al. (1999). Additionally, we used a two-way ANOVA (random vs. sheep location, and season, as main effects) to compare distances from the Briggs Mine to sheep locations in mined and nonmined areas.

Home-range Analyses

Prior to calculating adaptive-kernel home ranges for each female sheep, we used CALHOME to estimate the parameter for the optimum smoothing of the 95% adaptive kernel for that sheep (Worton 1989, Kie et al. 1996). Next, we calculated 95% adaptive-kernel home ranges based on 60-120% (in increments of 10%) of that smoothing parameter; the value that minimized the least squares cross-validation score for each individual data set was then used as the smoothing parameter for calculating 95, 50, and 10% adaptive-kernel home ranges for that animal (Kie et al. 1996). We considered the 10% adaptive-kernel home range to be the center of activity for a particular sheep; we then used the GIS to measure distances from that centroid of the home range to permanent springs and roads.

To determine if we had an adequate number of locations to estimate home ranges for each mountain sheep, we first plotted the area within 95% adaptive-kernel home range against cumulative sample size, and then estimated the sample size necessary to compute the home range using the nonlinear procedure in SPSS (SPSS Inc. 1993): home-range size = $A(1 - e^{bn})$, where A is the asymptote of the equation, e is the base of the natural log, n is the sample size, and b is a constant. Data sets that did not attain 90% of that asymptotic value were eliminated from further analyses (Nicholson et al. 1997).

We tested hypotheses regarding home-range size with data from seasons combined, because of inadequate numbers of locations within seasons. We analyzed size of home ranges using a two-sample *t*-test, and a one-way analysis of covariance (ANCOVA) with area as the main effect, and distances from the center of activity to nearest spring and road as covariates. We used the multi-response permutation procedure (MRPP) of BLOSSOM statistical software (Slauson et al.[4] 1991) to test our assumption that sheep assigned to mined and nonmined populations inhabited unique geographic areas.

Foraging Behavior

We collected data on head position (i.e., up vs. down) of female mountain sheep fitted with activity collars (mined, $n = 8$; nonmined, $n = 10$) via two remote recording stations (Receiver Model 2100, Data Logger Model DCC-5400, Advanced Telemetry Systems, Isanti, Minnesota) deployed from March 1996 to September 1997. The telemetry frequency of each sheep was scanned at 15-min intervals until either that frequency was received and a pulse rate could be ascertained, or for a maximum of 1 min. If that frequency was not received within 1 min., the datalogger proceeded to the next frequency in its memory and repeated the previously described process. Each time a frequency was received and a corresponding pulse rate was determined for that frequency, that observation (i.e., an instantaneous scan; Altmann 1974) was coded as either head-up or head-down; the resulting data were then pooled by individual animal at 1-hour intervals. We used a two-sample *Z*-test for proportions (Remington and Schork 1970) to evaluate the ability of the recording system to correctly quantify the proportion of time an animal spent with its head in a particular position, and to compare the proportion of time spent feeding with the position of the head. We further evaluated bias of the recording system following the methods of Hansen et al. (1992).

We conducted validation tests for concordance between direct visual observation and the recording system at the Bighorn Institute, Palm Desert, California, USA. One desert sheep was fitted with an activity collar (Advanced Telemetry Systems, Isanti, Minnesota), and the position of its head was recorded every 30 seconds using our electronic system. We conducted instantaneous-scan sampling (Altmann 1974) simultaneously with the collection of electronic data to serve as a measure of the "true" activity of the collared animal at that instant; we recorded head position (up or down) and the activity in which the animal was engaged (e.g., feeding, bedded, walking, etc.) at the same time the datalogger recorded its observation. We used data on head position to test the system for accuracy, whereas information on feeding was used to establish a relationship between head position and time spent feeding (Bradshaw et al. 1997). During validation of data collection by our remote system, the proportion of head-up and head-down observations recorded by the datalogger and the observer ($n = 1,277$) did not differ ($Z = 0.775$, $P = 0.441$); the recording system underestimated head-down positions by 1.2%. Additionally, no significant difference existed in the proportion of time in the head-down position (as indexed by the recorder) and the proportion of time spent foraging in direct observations ($Z = 0.163$, $P = 0.873$). Bias

associated with foraging was minimal with the electronic system, which overestimated observed foraging by 0.2%; consequently, we assumed that a head-down signal indicated a foraging animal, whereas a head-up signal was consistent with activities other than foraging.

Response to Blasting

Because we knew the days on which blasting occurred at the Briggs mine, we used days since blasting as our measure of response to that potential disturbance. We used a weighted mixed-model ANOVA (PROC MIXED; SAS Institute Inc. 1997) to test for the effect of blasting on head position (i.e., foraging vs. nonforaging); this model is appropriate when data contain both fixed and random components, and exhibit heterogeneous variances (SAS Institute Inc. 1997). Number of observations for each female during a particular day and hour were used to weight proportional data. Fixed effects were area, season, number of days since blasting (day of the blast, 1-3 days post-blasting, and >3 days post-blasting), and time of day (0100-0459, 0500-0859, 0900-1259, 1300-1659, 1700-2059, and 2100-0059 h). Individual animals (nested within area) were considered a random component, and were included to account for individual variation among animals; interactions incorporating that term also were considered random effects (SAS Institute Inc. 1997). The interaction season × time period × animals nested within area was used to test for differences between areas resulting from ecological processes (i.e., seasonal and diurnal patterns), whereas days since blasting × time period × animals nested within area was used to test for effects of blasting.

Forage Abundance and Quality

We quantified vegetation using step-point sampling along 100-m transects that were located randomly (Bowyer and Bleich 1984, Bleich et al. 1997) within mined and nonmined areas. We compared relative abundance between those areas using MANOVA and univariate *F*-tests. Samples of 12 forage species (five samples/species/area/month), consumed by mountain sheep (Bleich et al. 1992), were collected from July 1995 to June 1996 for analyses of percent crude protein (nitrogen × 6.25), in vitro dry matter digestibility (IVDMD), and moisture content following the methods of Bleich et al. (1992). Species of perennial forbs sampled were desert mallow, *Sphaeralcea ambigua*, desert trumpet, *Eriogonum inflatum*, and Rixford eriogonum, *E. rixfordii*. Perennial grasses were needlegrass, *Stipa speciosa*, and three-awn, *Aristida glauca*; representative species of shrubs included bedstraw, *Galium stellatum*, burro-weed, *Ambrosia dumosa*, desert holly, *Atriplex hymenelytra*, brittle bush, *Encelia farinosa*, Mormon-tea, *Ephedra nevadensis*, California buckwheat, *Eriogonum fasciculatum*, and mesquite, *Prosopis glandulosa*.

We analyzed IVDMD and moisture of perennial forbs and shrubs separately using three-way ANOVA (main effects were area, season, and forage class). Crude protein of perennial forbs and shrubs was analyzed with a three-way ranked ANOVA (Conover and Iman 1981) with the same factors. Because of nonconstant variances when grass

was incorporated into the overall model, that forage class was analyzed separately with a two-way ANOVA (area and season as main effects).

Diet Quality and Composition

We used a combination of telemetry (aerial and ground-based) and field observations to locate female groups for collection of fresh fecal pellets (i.e., <1-week-old) each month (June 1995-September 1997) within our study areas. We collected samples on approximately the same date each month, and stored them appropriately (Jenks et al. 1990) prior to conducting analyses. Because the California Department of Fish and Game had collared adult males ($n = 10$) in our study area as part of another project, we were able to use telemetry and direct observation to avoid areas inhabited predominantly by males (Bleich et al. 1997), thereby avoiding biases that might be introduced if we included their samples in our analyses. We attempted to collect ≥ 5 pellet groups (25 pellets per group) from each area each month. We determined percent fecal nitrogen for each sample as described previously for forage samples; this measure provided an index to diet quality (Bleich et al. 1997).

We used composited fecal samples (Bleich et al. 1997) collected between June 1995 and August 1996 to index composition of diets. Species of plants in fecal samples were determined at the Forage Analysis Laboratory, University of Arizona, with the microhistological technique described by Sparks and Malechek (1968). Plant fragments were categorized as perennial forbs, perennial grasses, shrubs, or succulents for statistical analyses (Bleich et al. 1997). Diet composition was analyzed with a two-way MANOVA with forage classes as dependent variables and area and season as main effects, whereas fecal nitrogen was evaluated with a two-way ANOVA with area and season as main effects.

Additional Analyses

We indexed relative abundance of carnivores on each area by noting when they were encountered during helicopter flights, and by collecting carnivore feces in the field. Feces were enumerated and pooled within each area, and feces per kilometer of line transect for each area was compared with a *t*-test (Bleich et al. 1997).

We compared the ratios of young to adult females observed during fieldwork and helicopter surveys using a binomial approach (Bowyer 1991). We calculated 95% *CI* for estimates, and compared ratios between areas for a particular period with the 95% *CI*; where *CI* overlapped, we assumed the ratio of young to adult females did not differ during that period (Bowyer 1991).

When multi-factor ANOVA was employed, all individual factor levels and their interactions were evaluated; significant models ($P \leq 0.05$) were explored further with Tukey's honestly significant difference (HSD) to determine where differences occurred. We analyzed data using the software PC SAS (SAS Institute Inc. 1997) and SPSS (Statistical Package for the Social Sciences 1993). We used a Bonferroni correction (Kleinbaum et al. 1988) when conducting multiple comparisons. We examined

assumptions of each statistical test and transformed data as necessary to meet those assumptions. Bivariate correlations were evaluated with a Pearson product-moment correlation (Zar 1984). An $\alpha = 0.05$ was adopted for all tests. Unless otherwise noted, we present means and standard errors for descriptive statistics.

RESULTS

Capture and Aerial Telemetry

We captured 8 female sheep (5 in 1995 and 3 in 1996) in the mined area, and 11 (8 in 1995 and 3 in 1996) in the nonmined area; no deaths of animals occurred during our capture efforts. During 45 h of extensive capture and survey efforts with a helicopter, we observed only 22 individual adult females within the specific area encompassing both study sites (Fig. 2), of which 19 (86%) were radiocollared during some portion of this study. We conducted 70 telemetry flights during June 1995-October 1997; female sheep were located 653 times (340 on the mined and 313 on the nonmined areas); the number of locations/female was 48.6 ± 5.8 on the mined and 39.1 ± 3.9 on the nonmined area. Additionally, our initial assignment of females to discrete populations (Fig. 2; i.e., mined or nonmined) was supported by their differing spatial distributions (MRPP; $\delta = -124.340, P < 0.001$).

Habitat Selection

Number of independent locations per female sheep in our analyses was 43.3 ± 4.4 in the mined and 34.1 ± 3.8 in the nonmined areas. The logistic-regression model, which exhibited good fit ($X^2 = 4.54, P = 0.85$), identified four variables as useful in distinguishing between random sites and those used by female sheep: percentage of the mixed-woody-scrub plant community ($X^2 = 108.80, P < 0.001$), elevation ($X^2 = 17.88, P < 0.001$), percent slope ($X^2 = 22.89, P < 0.001$), and percent visibility ($X^2 = 3.03, P < 0.001$; Table 1). Females from mined and nonmined areas, however, did not differ in how they used habitat (MANOVA, $F_{8,70} = 0.988, P = 0.452$). When compared to random locations, both groups selected sites in the mixed-woody scrub plant community at lower elevations, on steeper slopes, and with less visibility (Fig. 4).

Distance to permanent springs did not enter the logistic-regression model; however, because of an *a priori* hypothesis concerning its importance to desert sheep, we used a two-way ANOVA (area and season as main effects) to address that variable. When data from both areas were pooled, distance from water to random sites and to those used by female sheep differed significantly ($F_{5,1151} = 4.34, P = 0.013$). After controlling for availability of water, female sheep from the mined area were significantly nearer water than those from the nonmined area during autumn ($F_{1,364} = 9.27, P = 0.002$). During spring and summer, however, distance to water was not significantly different between areas ($F_{1,232} = 2.69, P = 0.102$ and $F_{1,544} = 0.06, P = 0.799$, respectively). In general, females from both areas were nearer water than were random locations during summer (Table 1).

Within the mined area, significant differences occurred in the spatial distribution

Table 1. Habitat characteristics of random locations and of female mountain sheep from mined and nonmined areas in the Panamint Range, Inyo County, California, USA, during 1995-1997. Sample sizes are in parentheses.

	Area													
			Mined						Nonmined					
	Random (576)		Spring (65)		Summer (150)		Autumn (88)		Spring (53)		Summer (124)		Autumn (96)	
Habitat variables	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Dist. to springs (m)	1,925	40	2,060	118	1,608	83	1,922	97	2,113	133	1,724	84	2,238	89
Dist. to roads (m)	1,846	50	1,501	112	1,579	56	1,814	83	1,508	133	1,726	89	1,363	93
Dist. to Briggs Mine (m)	12,059	367	2,975	326	2,091	167	3,267	340	19,698	651	21,092	320	20,960	436
Sin aspect[a]	-0.43	0.02	-0.52	0.03	-0.47	0.02	-0.43	0.04	-0.48	0.04	-0.40	0.03	-0.45	0.03
Cos aspect[b]	-0.07	0.02	-0.04	0.04	-0.08	0.04	-0.07	0.05	-0.13	0.06	-0.21	0.04	-0.10	0.05
Slope (%)[c]	26.1	0.3	27.5	0.8	30.5	0.5	28.6	0.6	27.1	0.7	29.9	0.6	26.7	.0.7
Terrain index[d]	526	12	664	31	545	21	573	30	575	39	498	20	534	29
Elevation (m)[c]	1,324	25	1,128	39	1,164	26	1,256	27	1,112	40	1250	27	1,157	37
Visibility (ha)[c,e]	0.61	0.02	47.0	3.4	51.4	2.0	45.9	2.8	49.7	3.8	59.6	2.6	55.1	3.1
Creosote-bush	7.3	1.0	3.7	2.2	0.9	0.7	0.0	0.0	3.9	2.4	0.9	0.6	2.9	1.6
Juniper-woodland	11.6	1.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	0.8	1.0	1.0
Mixed-woodyscrub[c]	79.56	1.6	95.5	2.6	99.1	0.7	100	0.0	96.1	2.4	98.4	1.0	96.0	1.9

[a]N-S aspects.

[b]E-W aspects.

[c]Variables selected by logistic regression model for differentiating random locations from those used by female sheep.

[d]SD slope times angular deviation of aspect (Nicholson et al. 1997).

[e]Viewshed analysis from ARC/INFO; area (ha) visible to a sheep (1 m in height) to a maximum distance of 1000 m based on topographic relief.

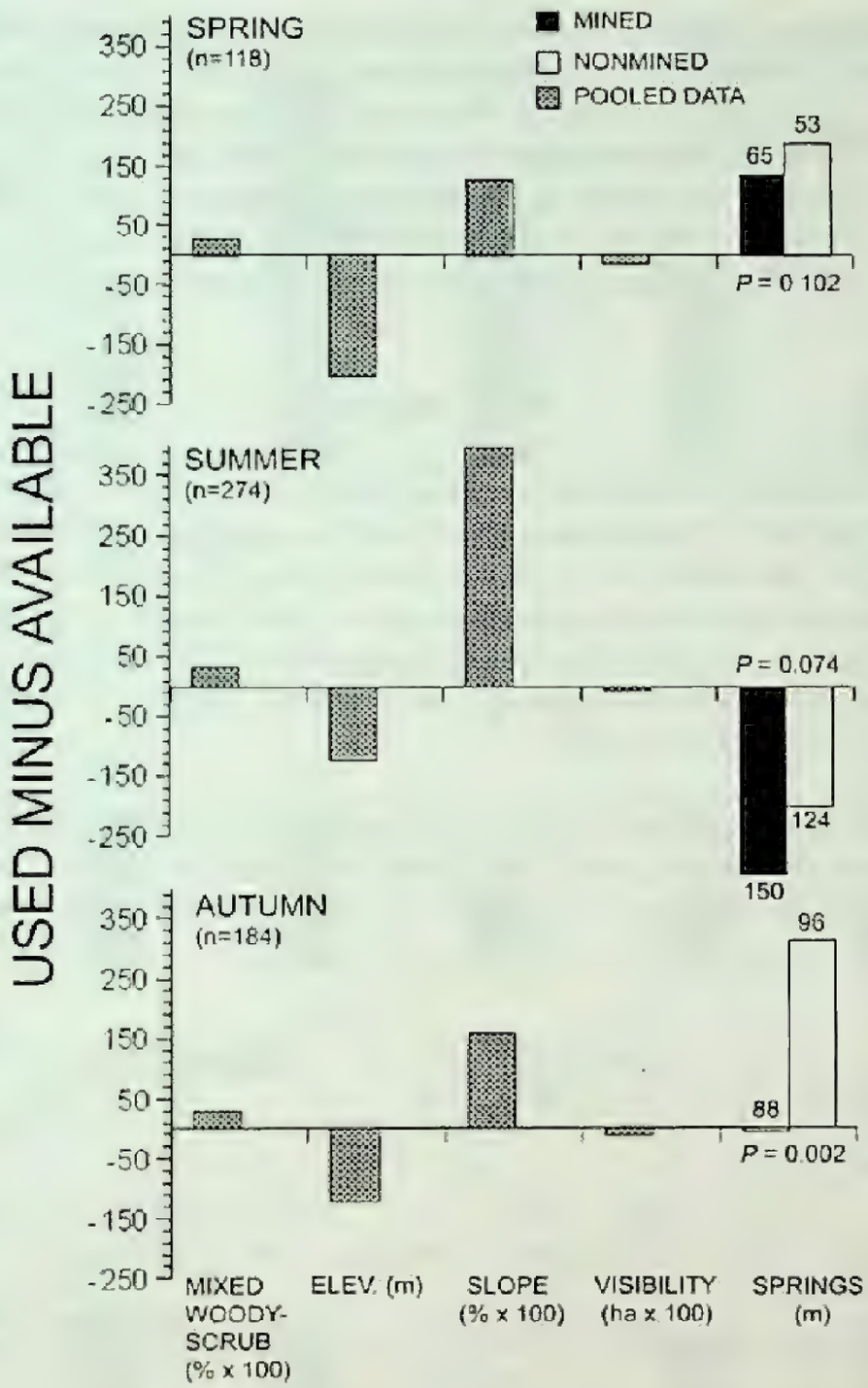

Fig. 4. Selection (used minus available) of habitat variables by female mountain sheep in the Panamint Range, Inyo County, California, USA, during 1995-1997. Mixed woody-scrub, elevation, slope, and visibility were significant variables in a logistic regression model ($X^2 = 4.54$, $P = 0.85$) for differentiating sheep locations from random locations. P-values for distance to springs are from ANOVA, and numbers above or below bars represent number of sheep locations used in that analysis.

of sheep and random locations relative to the Briggs Mine (two-way ANOVA, $F_{5,605}$ = 19.00, P < 0.001); females were consistently nearer the mine than were random locations (ANOVA, $F_{1,605}$ = 67.09, P < 0.001). Moreover, there were significant differences between seasons in the distances that sheep occurred from the mine ($F_{2,302}$ = 6.68, P < 0.001); female sheep were closer to the mine during summer (2,091 ± 166 m) than in autumn (3,267 ± 340 m), whereas during spring they were at an intermediate distance from the mine (2,975 ± 326 m); distance from the mine in spring did not differ from that in summer.

Home Range

We determined that a mean minimum sample of 26.7 ± 2.4 telemetry locations in the mined (n = 7) and 26.5 ± 2.6 in the nonmined (n = 8) areas were required for home range estimation; four individuals lacked an adequate sample and were eliminated from our analyses. Mean sizes of annual home ranges did not differ significantly between areas (Table 2). Moreover, sizes of home ranges did not differ after controlling for effects of distance to the nearest permanent spring (ANCOVA, $F_{1,12}$ = 0.482, P = 0.501), or road (ANCOVA, $F_{1,12}$ = 0.325, P = 0.579).

Table 2. Size of annual home ranges (ha) of female mountain sheep from mined and nonmined areas in the Panamint Range, Inyo County, California, USA, during 1995-1997. P-values from two-sample t-tests are for within-row comparisons of mined versus nonmined areas.

	Area						
	Nonmined (n = 8)			Mined (n = 7)			
Home Range Model	$\bar{x}$	SE	CV	$\bar{x}$	SE	CV	P
Adaptive Kernel							
95%	6,926	834	34	6,222	806	34	0.557
50%	1,230	193	44	954	118	33	0.260
Min. Convex Polygon							
95%	4,006	445	31	3,467	396	30	0.395

Foraging Behavior

We collected 10,241 and 7,023 h of data on head position (hereafter foraging, or foraging activity) for sheep inhabiting the mined (n = 8) and nonmined (n = 10) areas, respectively (Fig. 5). Number of days since blasting occurred had a significant effect on the proportion of time that females spent foraging ($F_{4,334}$ = 17.68, P < 0.001). When a reduced model (all main effects and the interaction days since blasting × time period × animals nested within area) was conducted by season, the three-way interaction was highly significant during all 3 seasons. That effect was greatest in summer (Z = 8.07, P < 0.001), intermediate during spring (Z = 7.49, P = 0.001), and smallest during autumn (Z = 6.81, P = 0.001).

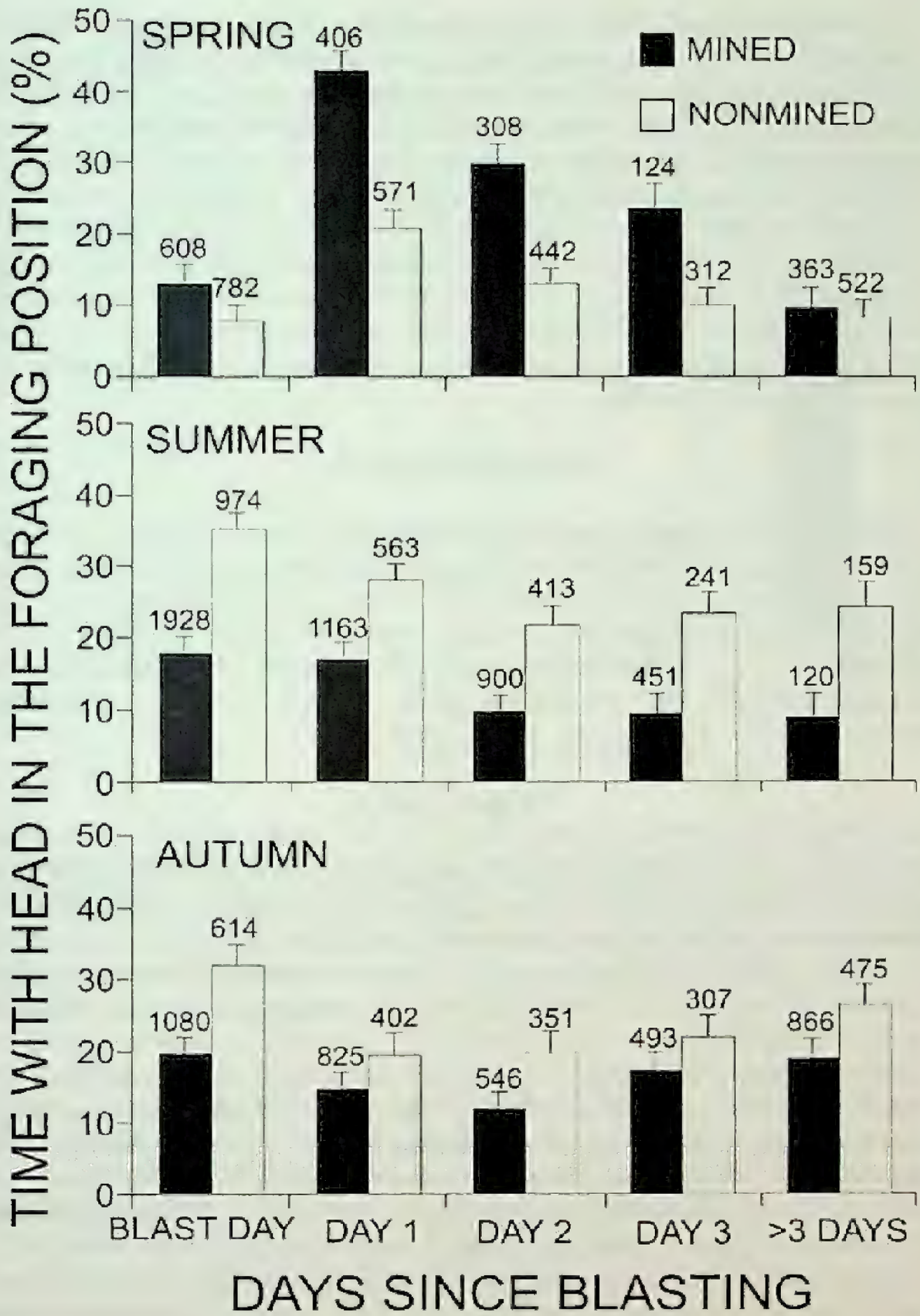

Fig 5. Percent time with the head in a foraging position (indexed by tip-switch collars) for female mountain sheep from mined and nonmined areas in the Panamint Range, Inyo County, California, USA, during 1996-1997. Value of bars (+ 1 *SE*) are least-squares means from mixed-model ANOVA (PROC MIXED; SAS 1997); numbers above bars are total hours of data collected for that bar.

We further examined number of days between blasts for the mined area with a two-way MANOVA, with days between blasting and amount of explosives per blast as dependent variables, and season as the class variable. We noted significant differences among seasons ($F_{4,400} = 3.58, P = 0.007$), which were not the result of differences in the amount of explosives used per blast during spring (20,462 ± 1,362 tons), summer (23,204 ± 905 tons), or autumn (22,020 ± 1,280 tons) ($F_{2,201} = 1.61, P = 0.202$), but rather number of days between blasts ($F_{2,201} = 5.76, P = 0.004$). There were significantly fewer days between blasts during summer (1.5 ± 0.2) and spring (2.2 ± 0.3) than during autumn (3.0 ± 0.4) ($F_{2,203} = 5.76, P = 0.004$). There were 3,397,878 metric tons of ore removed from the pit during spring, 4,785,498 in summer, and 2,552,166 during autumn. Amount of ore hauled each month was positively correlated with tons of explosives used during that month ($r^2 = 0.83, P < 0.001$).

Forage Abundance

We quantified vegetation on 24 and 36 transects in the mined, and nonmined areas, respectively (Fig. 6), and documented significant differences between areas in types of ground cover ($F_{1,58} = 12.21, P < 0.001$). Transects on the nonmined area were characterized by more annuals ($F_{1,58} = 48.55, P < 0.001$), forbs ($F_{1,58} = 7.16, P = 0.009$), and bare ground ($F_{1,58} = 8.41, P = 0.005$) than those in the mined area, whereas abundance of succulents ($F_{1,58} = 3.79, P = 0.056$) and shrubs ($F_{1,58} = 0.00, P = 0.99$) did not differ between areas.

Forage Quality

We collected 1,152 forage samples within mined ($n = 578$) and nonmined ($n = 574$) areas. No significant differences occurred in moisture content of perennial forbs and shrubs between areas, but differences existed among seasons ($F_{11,191} = 1.88, P = 0.044$), with shrubs possessing a higher moisture content than perennial forbs during spring ($F_{3,77} = 3.35, P = 0.023$; Fig. 7). Similarly, there were no significant differences between areas in IVDMD of perennial forbs or shrubs ($F_{1,191} = 2.90, P = 0.090$); nevertheless, IVDMD of perennial forbs and shrubs differed significantly among seasons ($F_{1,191} = 24.62, P < 0.001$). Shrubs had consistently higher IVDMD than did perennial forbs during spring ($F_{1,77} = 22.15, P < 0.001$) and summer ($F_{1,63} = 6.30, P = 0.015$), but that relationship was not as apparent during autumn ($F_{1,49} = 3.30, P = 0.075$). Conversely, when an overall model considered crude protein of perennial forbs and shrubs, there were no area or seasonal effects ($F_{11,191} = 0.90, P = 0.546$).

When perennial grasses were analyzed separately, there were significant differences in protein between seasons ($F_{5,23} = 7.52, P < 0.001$); further examination revealed differences were attributable to perennial grasses having higher protein content during spring in both mined ($F_{2,11} = 6.55, P = 0.017$) and nonmined ($F_{2,11} = 9.87, P = 0.005$) areas. Likewise, significant differences occurred in IVDMD ($F_{5,23} = 6.42, P < 0.001$) and moisture content ($F_{5,23} = 4.52, P = 0.008$) of perennial grasses from mined and nonmined areas. Again, differences were driven largely by the effects of spring; IVDMD was

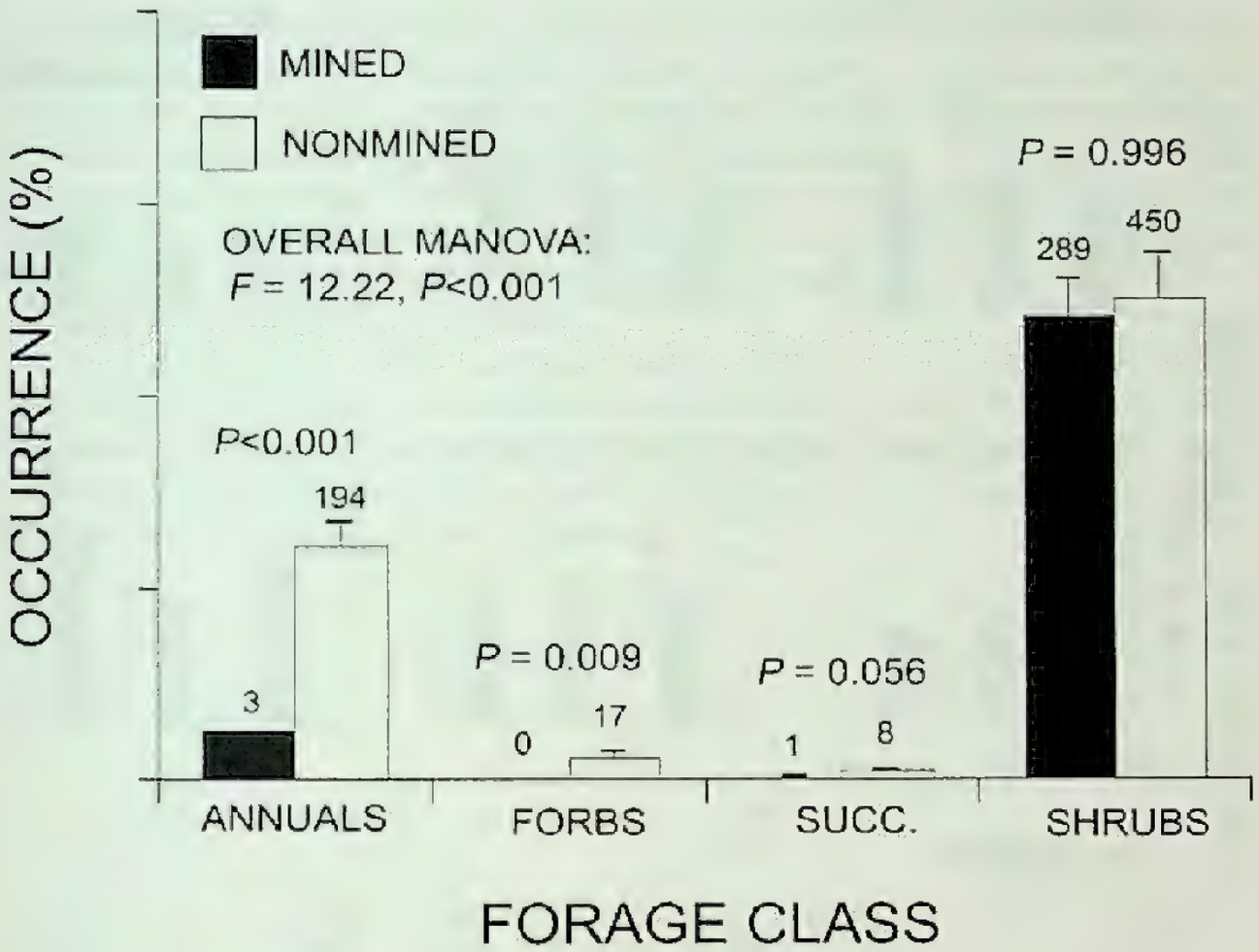

Fig. 6. Mean (+ 1 *SE*) vegetative cover (%) in habitats used by radiocollared female mountain sheep from mined (n = 24 transects) and nonmined (n = 36 transects) areas in the Panamint Range, Inyo County, California, during 1995-1997. Numbers above bars indicate occurrence (i.e., "hits") of that type of vegetation on all transects, and P-values are from univariate F tests. Grasses were not detected on transects in either area, and percentage of bare ground was significantly greater (P = 0.005) in mined (87%) than the nonmined area (81%).

significantly higher ($F_{1,5} = 29.58, P = 0.005$) on the mined than the nonmined area ($\bar{x}$= 66.4 $\pm$ 4.7 and $\bar{x}$ = 42.5 $\pm$ 5.9%, respectively) during spring. Moisture content of perennial grasses did not differ between areas ($F_{2,23}$ = 2.18, P = 0.157), but was significantly higher in both areas during spring than in autumn ($F_{2,11} = 7.39, P = 0.013$). Overall, there was a clear trend in both areas for increased quality of forage among all classes during spring (Fig. 7).

Diet Quality and Composition

We collected 175 individual fecal groups from the mined area and 184 from the nonmined area for assessing quality of diet. There were significant area and seasonal effects on quality of diets (two-way ANOVA; $F_{13,358} = 32.18, P < 0.001$; Fig. 8). Female

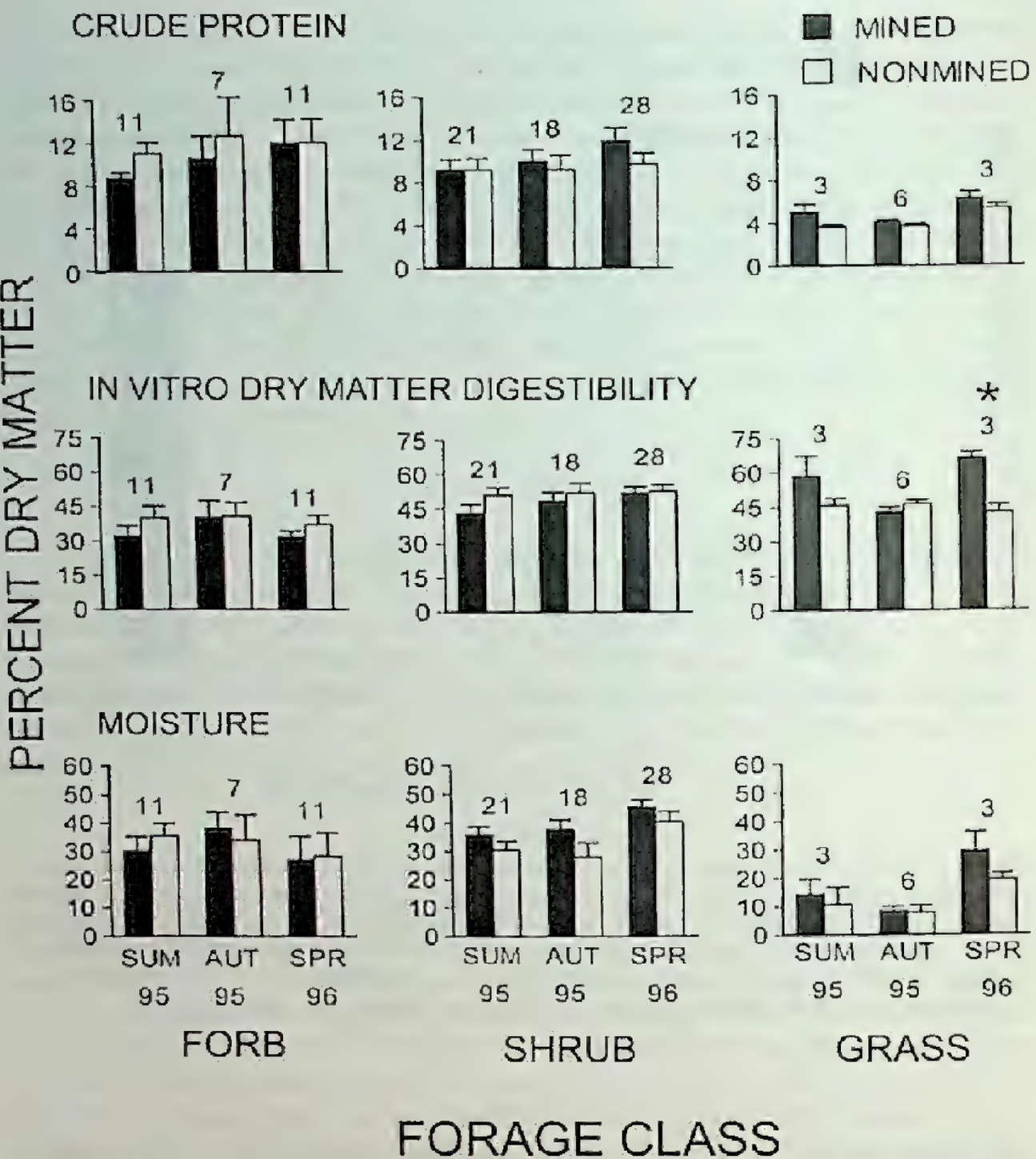

Fig. 7. Percent dry matter crude protein (top), *in vitro* dry matter digestibility (middle), and moisture (bottom) content of forage classes ($\bar{x} \pm SE$) eaten by mountain sheep in the Panamint Range, Inyo County, California, during 1995-1996. Asterisk indicates significant difference ($P < 0.01$) between areas and numbers above bars represent sample size from each area in the comparison.

sheep on the nonmined area had consistently higher levels of fecal nitrogen than conspecifics in the mined area (Fig. 8). Moreover, diet quality was highest during spring (Fig. 8), which was consistent with forage classes having higher levels of crude protein, moisture, and increased digestibility during that season (Fig. 7). Although level of tannins in shrubs may have increased fecal nitrogen, this is unlikely to have occurred because there was not a consistent or significant correlation between fecal nitrogen and the amount of shrubs in the diets of sheep from mined ($r^2 = 0.882$, $P = 0.118$), or nonmined ($r^2 = -0.268$, $P = 0.732$) areas.

No significant differences occurred between areas in the proportions of forage classes in the diet of female sheep (two-way MANOVA, $F_{8,32} = 1.27$, $P = 0.292$). During all seasons, shrubs were the most prevalent vegetation type in diets of females from both areas (Table 3). Overall, diets of female sheep contained an average of 55% shrubs, 30% forbs, 11% succulents, and 4% grasses.

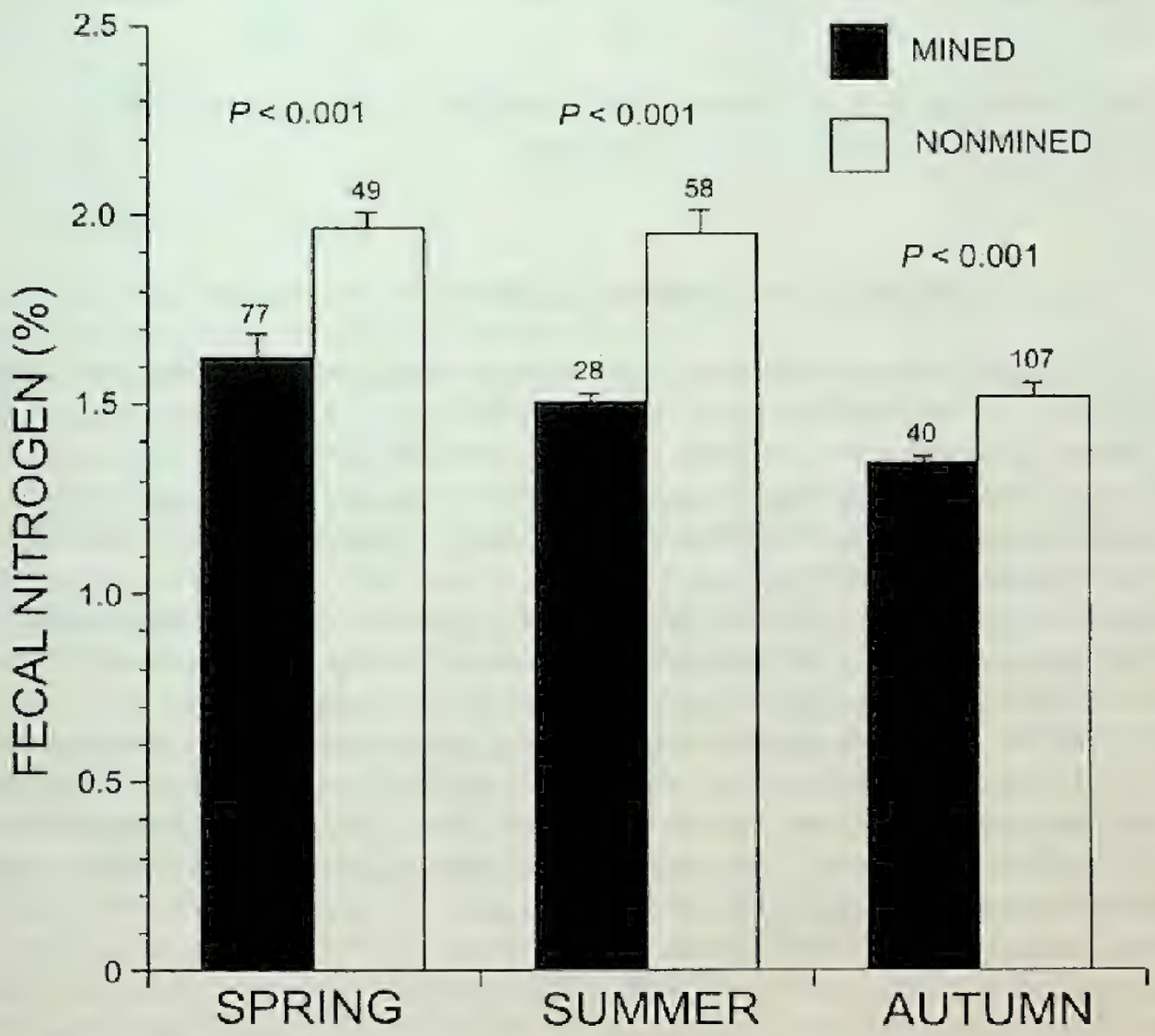

Fig. 8. Mean percent fecal nitrogen of female mountain sheep from mined and nonmined areas in the Panamint Range, Inyo County, California, during 1995-1997. Numbers above bars indicate sample size, bars represent 1 *SE*, and *P*-values are from two-sample *t*-tests.

Table 3. Percent of forage classes in the diets of female mountain sheep indexed from microhistological analyses of their feces, from mined and nonmined areas in the Panamint Range, Inyo County, California, USA, during 1995-1996.

	Percentage of forage class[a]							
	Forb		Grass		Shrub		Succulent	
Area	$\bar{x}$	*SE*	$\bar{x}$	SE	$\bar{x}$	SE	$\bar{x}$	SE
Spring								
Mined (3)[b]	34.4	2.3	1.1	0.6	53.5	2.4	11.1	0.7
Nonmined (3)	26.4	8.3	4.1	0.6	53.9	8.9	15.6	1.0
Summer								
Mined (7)	27.1	3.8	4.5	1.4	59.3	5.1	9.2	1.7
Nonmined (6)	31.0	5.7	5.9	1.0	52.9	4.9	10.3	2.4
Autumn								
Mined (2)	35.4	5.5	1.5	1.5	48.9	3.4	14.3	0.6
Nonmined (4)	31.3	5.2	1.8	1.4	57.5	8.1	9.4	3.8

[a]Overall MANOVA (area and season) was not significant ($F_{8,32} = 1.27$, $P = 0.292$).
[b]Number of composite fecal samples for that season.

Mortality and Indices to Predator Abundance

There were seven mortalities of female sheep in the nonmined area, and two in the mined area. In the nonmined area, two mortalities were attributed to predation by mountain lions, one fell to its death, and causes of death for the other four could not be ascertained. Similarly, causes of mortality for two female sheep in the mined area could not be determined. No differences existed in the ratio of young to adult females between mined and nonmined areas (Table 4).

From July 1995 to September 1997, we sampled transects totaling 24 and 34 km in length for carnivore feces from the mined and nonmined areas, respectively. When feces encountered on transects were pooled by area, there was no significant difference ($t_{22} = -1.91$, $P = 0.077$) in the number of feces/km between mined (1.6 ± 0.3) and nonmined (1.0 ± 0.1) sites. No carnivores were sighted during the course of fieldwork on either study area; however, most work was performed during the day. Additionally, no carnivores were observed on the study area while conducting captures of sheep (~45 h of helicopter flight-time).

DISCUSSION

Collared mountain sheep constituted 86% of the adult female sheep observed in our two study areas. Overall, we observed few effects that we could attribute to mining on the demography or other characteristics of populations of female sheep inhabiting mined and nonmined areas. For instance, proportion of young to adult females was not

Table 4. Young and adult female mountain sheep observed during aerial surveys near mined and nonmined areas in the Panamint Range, Inyo County, California, USA, during 1995-1997.

Date Area	Young (*n*)	Females (*n*)	P_i[b]	95% *CI* (P_i)
Jun 1995[a]				
Mined	5	7	0.416	0.132-0.700
Nonmined	6	8	0.428	0.290-0.909
Oct 1995[a]				
Mined	1	4	0.200	0.000-0.551
Nonmined	6	10	0.375	0.138-0.612
Jun 1996[a]				
Mined	4	8	0.333	0.117-0.549
Nonmined	5	14	0.260	0.007-0.513
Jan 1997[a]				
Mined	3	4	0.428	0.054-0.802
Nonmined	2	4	0.666	0.281-1.000
Oct 1997[a]				
Mined	3	4	0.428	0.054-0.802
Nonmined	3	6	0.333	0.019-0.647

[a]Proportion of young to adult females did not differ between areas during that survey as indicated by overlap of 95% confidence intervals.
[b]Ratio and confidence intervals (CI) determined following methods of Bowyer (1991).

different between areas during any of the 3 years we monitored those subpopulations (Table 4). Moisture content, crude protein, and IVDMD of forage classes were highest for both areas in spring, intermediate in summer, and lowest during autumn (Fig. 7). Those outcomes are consistent with patterns in the quality of forage observed for other ranges inhabited by sheep in desert environments (Bleich et al. 1997, Krausman et al. 1989). Digestibility of grass was highest in the mined area during spring, but that was the only difference we detected in forage quality between areas (Fig. 7). We observed no differences in composition of diets of female sheep from mined and nonmined areas, although differences existed between forage classes consumed across seasons (Table 3). The former outcome is probably the result of the similarity in quality (Fig. 7) and availability of forage species (i.e., shrubs) between sites (Fig. 6), whereas the latter is likely a consequence of how desert plants with differing life-history strategies respond to variation in precipitation (Beatley 1974). Thus, female sheep from both areas used forage classes differentially, depending on seasonal quality and availability. Differences in availability of forage (as indexed by vegetative cover) between areas were minimal and limited to percentage of annual plants and perennial forbs (Fig. 6).

Size of annual home ranges for females did not differ significantly between areas (Table 2). Indeed, there was substantial individual variation and no clear pattern in

those data (Table 2). Berger (1991) suggested that use of steep rocky habitats was more pronounced for post-parturient female mountain sheep when compared with those without young. In contrast, females without offspring were more likely to forage in areas away from escape terrain, where quality of forage was better, but perceived risk of predation higher (Berger 1991)—those factors could result in large home ranges for females without young. Thus, variability in sizes of home range in our study may have occurred because not all collared females had young at heel, a hypothesis we could not test because we were unable to ascertain the reproductive status of every female during aerial telemetry flights. Nonetheless, the large proportion of the population that was collared and the lack of a difference in ratios of young to adult females between study areas make this interpretation unlikely.

Females from both areas selected sites with more mixed woody-scrub, lower elevations, steeper slopes, and less visibility than at random locations during all seasons (Fig. 4, Table 1). Several researchers (Berger 1991, Bleich et al. 1997, and others) have reported that female sheep consistently used steep and rugged terrain that was close to water. That strategy likely represents a tradeoff between decreased forage quality in steep rocky habitats and decreased rates of predation on neonates (Berger 1991, Bleich et al. 1997, Rachlow and Bowyer 1994, 1998). Following precipitation in early spring, the proportion of grasses increased in the diets of sheep from the nonmined area, but not for those in the mined area (Table 3). That females in the nonmined area apparently increased their use of that resource is consistent with other studies that noted an increase in the consumption of graminoids by mountain sheep following periods of precipitation (Wehausen and Hansen 1988, Berger 1991). Berger (1991) proposed that pre-parturient females that foraged at low elevations on open slopes traded an increased risk of predation for the opportunity to maximize nutrient intake, a hypothesis supported by research on other ungulates (Bowyer et al. 1999, Kie 1999, Barten et al. 2001). Females from the nonmined area had higher levels of fecal nitrogen during spring than did those from the mined area (Fig. 8); such an outcome would be consistent with the aforementioned strategy suggested by Berger (1991). Although females from both areas used areas lower in elevation than random locations, females from the mined area may have been reluctant to forage on the lowest elevation slopes adjacent to the mine during spring because of the proximity of those sites to activities associated with the mine.

Patterns of foraging were not similar between subpopulations of female sheep (Fig. 5), and interpretation of those results during spring was not straightforward. If females in the mined area were precluded from foraging on graminoids during spring by mining activities, they may have had to forage more intensively (i.e., spent more time foraging) in steep areas where forage quality was lower (Bleich et al. 1997). Indeed, quality of diet was lower for female sheep in the mined area during spring (Fig. 8). Although mountain sheep may habituate to human-caused disturbances (Morgantini and Worbets 1988), those ungulates have been reported to avoid areas where disturbance was extreme (Leslie and Douglas 1980, Berger 1991, Stockwell et al. 1991), and did not appear to habituate to extreme disturbances such as helicopter overflights (Bleich et al. 1994).

Availability and juxtaposition of water within each area may be the most parsimonious

explanation for differences we observed in foraging ecology between groups of female mountain sheep inhabiting mined and nonmined areas during summer. Indeed, females from both areas were nearer water during summer than other seasons (Table 1). Sources of permanent water were fewer and more dispersed in the mined area. Moreover, female sheep in that area relied almost exclusively on Redlands Spring, which was adjacent to the mine, to meet their metabolic needs for water during summer. In contrast, females in the nonmined area used several springs. Turner[5] (1973) reported that daily needs for water for desert mountain sheep was about 4% of their body mass, and that this amount could not be obtained from forage during the hot summer. Desert sheep have been reported to inhabit ranges without sources of perennial water (Krausman et al. 1985*b*); nevertheless, that result would not preclude water affecting the distribution of sheep in other areas. For instance, free water strongly affected the distributions of other ungulates, even in situations where water was abundant and therefore could not be limiting those populations (Bowyer 1981, Stewart et al. 2002).

Alderman et al. (1989) reported that in the Little Harquahala Mountains, Arizona, where permanent water was not available, desert sheep likely met their metabolic needs by using pools of free-standing water that accumulated in depressions in the substrate after occasional thunderstorms. During our investigation, however, summer thundershowers were rare (Oehler et al. 2003); the lack of summer rain likely would have limited areas that could be used by desert sheep during that season. As a consequence of their reliance on Redlands Spring during summer, female mountain sheep in the mined area also were closest to the Briggs Mine during that season (Table 1).

Several studies have reported that the magnitude of a response to a disturbance is a function of the proximity to the stimulus (MacArthur et al. 1982, Stockwell et al. 1991, Bleich et al. 1994). Stockwell et al. (1991) concluded that mountain sheep in the Grand Canyon, Arizona, foraged more efficiently as distance from helicopter disturbance increased. Similarly, MacArthur et al. (1982) reported mountain sheep in Alberta, Canada, exposed to low-flying aircraft (90-250 m), exhibited a 3.5-fold increase in heart rate over those exposed to high-flying aircraft (>400 m). Results from our analyses of foraging behavior and diet quality suggest that female sheep in the mined area were disturbed by activities associated with the Briggs Mine during summer. That outcome is consistent with the interval between blasting being shortest during summer (1.5 $\pm$ 0.2 days). Moreover, amount of ore hauled (an index to vehicle activity) from the mine pit also was highest during summer, and was strongly correlated with the amount of explosives used at the mine.

Patterns of decreased foraging by mountain sheep in the mined area during summer and autumn (Fig. 5) may have been the result of those females spending more time vigilant and, concomitantly, less time foraging (Berger 1991, Stockwell et al. 1991). That conclusion is in keeping with sheep in the mined area obtaining lower-quality diets during summer than sheep from the nonmined area (Fig. 8) as a consequence of decreased foraging efficiency (Berger 1991, Stockwell et al. 1991, Molvar and Bowyer

[5]Turner, J. C. 1973. Water, energy and electrolyte balance in the desert bighorn sheep *Ovis canadensis*. Dissertation, University of California, Riverside, USA.

1994, Bowyer et al. 2001). Likewise, summer was when the greatest disparity in quality of diets of sheep occurred (Fig. 8). Differences in levels of foraging during autumn were similar to those of summer for sheep in the mined area (i.e., consistently lower than the nonmined area; Fig. 5); however, interpretation is less clear than for summer.

During autumn, as daytime temperatures (Oehler et al. 2003) and metabolic needs for water decreased, levels of foraging were most similar between areas, and may have been an outcome of sheep from the mined area moving away from Redlands Spring (Table 1) and, consequently, away from the mine. Moreover, an increased interval between blasts at the mine, and subsequently less ore being removed during autumn than summer, may have ameliorated disturbances to sheep. Quality of forage was lowest during autumn (Fig. 7); however, decreased water requirements (Turner[5] 1973) associated with lower temperatures probably allowed sheep in the mined area to venture further from Redlands Spring in search of higher-quality forage. We hypothesize that the resultant increase in foraging by sheep in the mined area during autumn (Fig. 5), and the small difference in quality of diets between areas (Fig. 8), are the result of those animals moving away from the area of disturbance as physiological demands for water became less severe.

A combination of four factors offers the best explanation for our results: 1) proximity of Briggs Mine to the primary source of water used by females in the mined area; 2) a limited number of permanent sources of water in the mined area; 3) lack of rainfall during summer and autumn (Oehler et al. 2003); and 4) philopatric behavior of female mountain sheep (Geist 1968), which lessened the tendency to disperse from the disturbance caused by mining. In concert, those factors likely resulted in females remaining near the mine where disturbance was greatest, particularly during summer when metabolic needs for water also were highest. Leslie and Douglas (1980) reported that female mountain sheep in Nevada altered their watering patterns in response to construction activities near a primary source of water. In contrast, we did not observe such a result, presumably because of the high degree of fidelity of females to Redlands Spring and the scarcity of other nearby sources of free water.

Demographic consequences in response to disturbance are difficult to document because of high variability in observed population parameters (e.g., young to female ratios); therefore, less-direct measurements are of value for these types of investigations. For example, small changes in diet quality can result in important nutritional changes in females over time, and thereby affect subsequent reproductive efforts (White 1983). Also, nutrition, as affected by forage quality and efficiency of foraging (Berger 1979), can be linked to nutrition of females, and thereby survivorship of young ungulates (Keech et al. 2000). Such consequences can be especially important to desert mountain sheep, particularly in marginal environments, where recruitment is comparatively low (Rubin et al. 2000). If the outcomes we observed persist in the mined area, we hypothesize that reduced nutrient intake could have demographic consequences for that subpopulation.

Because observed differences were most pronounced during summer, a reduction of mining activities during that season may benefit sheep occupying areas near the mine. Most females have young at heel during spring, and shifting mining activity from

summer to spring might be detrimental. Increasing the interval between blasting, which would result in a decrease in overall mining activity during summer, may ameliorate the apparent effects of mining on sheep during that season. Intensity of mining, as indexed by days between blasting, amount of explosives used, and amount of ore hauled from the pit, was lowest during autumn. Although we have no direct data on levels of potential auditory disturbance from the sound caused by blasting, amount of explosives used undoubtedly provides an index to such stimuli. If a shift in mining activity is necessary to offset lost mine production in summer, we suggest that autumn is the best time for the concomitant increase in activities. Creation of additional sources of permanent water away from the mine also may help reduce reliance of females on Redlands Spring and, hence, reduce potential negative effects of mining on foraging behavior. We observed few effects of mining on demography or other characteristics of mountain sheep populations; however, we caution that because of the unique distribution of water in our study area, that our conclusions should not be generalized to other circumstances where the juxtaposition of critical resources might differ, thereby detrimentally affecting the demographics of mountain sheep.

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LITERATURE CITED

Ad Hoc Committee on Acceptable Field Methods. 1987. Acceptable field methods in mammalogy. Journal of Mammalogy 68(suppl.):1-18.

Agresti, A. 1990. Categorical data analysis. John Wiley & Sons, New York, New York, USA.

Alderman, J. A., P. R. Krausman, and B. D. Leopold. 1989. Diel activity of female desert bighorn sheep in western Arizona. Journal of Wildlife Management 53:264-271.

Altmann, J. 1974. Observational study of behavior: sampling methods. Behaviour 49:227-267.

Andrew, N. G., V. C. Bleich, and P. V. August. 1999. Habitat selection by mountain sheep in the Sonoran Desert: implications for conservation in the United States and Mexico. California Wildlife Conservation Bulletin 12:1-30.

Barten, N. L., R. T. Bowyer, and K. J. Jenkins. 2001. Habitat use by female caribou: tradeoffs associated with parturition. Journal of Wildlife Management 65:77-92.

Beatley, J. C. 1974. Phenological events and their environmental triggers in Mojave Desert ecosystems. Ecology 55:856-863.

Berger, J. 1978. Group size, foraging and anti-predator ploys: an analysis of bighorn sheep decisions. Behavioral Ecology and Sociobiology 4:91-100.

Berger, J. 1991. Pregnancy incentives, predation constraints and habitat shifts: experimental and field evidence for wild bighorn sheep. Animal Behaviour 41:61-77.

Bleich, V. C., R. T. Bowyer, D. J. Clark, and T. O. Clark. 1992. Analysis of forage used by mountain sheep in the eastern Mojave Desert, California. Desert Bighorn Council Transactions 36:41-47.

Bleich, V. C., R. T. Bowyer, A. M. Pauli, M. C. Nicholson, and R. W. Anthes. 1994. Mountain sheep *Ovis canadensis* and helicopter surveys: ramifications for the conservation of large mammals. Biological Conservation 45:1-7.

Bleich, V. C., R. T. Bowyer, and J. D. Wehausen. 1997. Sexual segregation in mountain sheep: resources or predation? Wildlife Monographs 134:1-50.

Bowyer, R. T. 1981. Activity, movement, and distribution of Roosevelt elk during rut. Journal of Mammalogy 62:574-582.

Bowyer, R. T. 1991. Timing of parturition and lactation in southern mule deer. Journal of Mammalogy 72:138-145.

Bowyer, R. T., and V. C. Bleich. 1984. Effects of cattle grazing on selected habitats of southern mule deer. California Fish and Game 70:240-247.

Bowyer, R. T., D. R. McCullough, and G. E. Belovsky. 2001. Causes and consequences of sociality in mule deer. Alces 37:371-402.

Bowyer, R. T., V. Van Ballenberghe, J. G. Kie, and J. A. K. Maier. 1999. Birth-site selection in Alaskan moose: maternal strategies for coping with a risky environment. Journal of Mammalogy 80:1070-1083.

Bradshaw, C. J. A., S. Boutin, and D. M. Hebert. 1997. Effects of petroleum exploration on woodland caribou in northeastern Alberta. Journal of Wildlife Management 61:1127-1133.

Conover, W. J., and R. L. Iman. 1981. Rank transformations as a bridge between parametric and nonparametric statistics. American Statistician 35:124-130.

DeForge, J. R., J. E. Scott, G. W. Sudmeier, R. L. Graham, and S. V. Segreto. 1981. The loss of two populations of desert bighorn sheep in California. Desert Bighorn Council Transactions 25:36-38.

Festa-Bianchet, M. 1988. Seasonal range selection in bighorn sheep: conflicts of forage quality, forage quantity, and predator avoidance. Oecologia 75:580-586.

Frid, A. 1997. Vigilance by female Dall's sheep: interactions between predation risk factors. Animal Behaviour 53:799-808.

Geist, V. 1968. On the interrelation of external appearance, social behavior and social structure of mountain sheep. Zeitschrift f"ur Tierpsychologie 25:199-215.

Hansen, M. C., G. W. Garner, and S. G. Fancy. 1992. Comparison of 3 methods for evaluating activity of Dall's sheep. Journal of Wildlife Management 56:661-668.

Hicks, L. L., and J. M. Elder. 1979. Human disturbance of Sierra Nevada bighorn sheep. Journal of Wildlife Management 43:909-915.

Jaeger, J. R., J. D. Wehausen, V. C. Bleich, and C. L. Douglas. 1993. Limits in the resolution of LORAN-C for aerial telemetry studies. Desert Bighorn Council Transactions 37:20-23.

Jenks, J. A., R. B. Soper, R. L. Lochmiller, and D. M. Leslie, Jr. 1990. Effects of exposure on nitrogen and fiber characteristics of white-tailed deer feces. Journal of Wildlife Management 54:389-391.

Johnson, R. A., and D. W. Wichern. 1988. Applied multivariate statistical analysis. Prentice-Hall, Englewood Cliffs, New Jersey, USA.

Keech, M. A., R. T. Bowyer, J. M. Ver Hoef, R. D. Boertje, B. W. Dale, and T. R. Stephenson. 2000. Life-history consequences of maternal condition in Alaskan moose. Journal of Wildlife Management 64:450-462.

Kie, J. G. 1999. Optimal foraging and risk of predation: effects on behavior and social structure in ungulates. Journal of Mammalogy 80:1114-1129.

Kie, J. G., J. A. Baldwin, and C. J. Evans. 1996. CALHOME: a program for estimating animal home ranges. Wildlife Society Bulletin 24:342-344.

Kie, J. G., R. T. Bowyer, M. C. Nicholson, B. B. Boroski, and E. R. Loft. 2002. Landscape heterogeneity at differing scales: effects on spatial distribution of mule deer. Ecology 83:530-544.

Kleinbaum, D. G., L. L. Kupper, and K. E. Muller. 1988. Applied regression analysis and other multivariate methods. Second edition. Duxbury Press, Belmont, California, USA.

Krausman, P. R., J. J. Hervert, and L. L. Ordway. 1985a. Capturing deer and mountain sheep with a net-gun. Wildlife Society Bulletin 13:71-73.

Krausman, P. R., S. Torres, L. L. Ordway, J. J. Hervert, and M. Brown. 1985b. Diel activity of ewes in the Little Harquahala Mountains, Arizona. Desert Bighorn Council Transactions 29:24-26.

Krausman, P. R., B. D. Leopold, R. F. Seegmiller, and S. G. Torres. 1989. Relationships between desert bighorn sheep and habitat in western Arizona. Wildlife Monographs 102:1-66.

Krausman, P. R., and R. C. Etchberger. 1995. Response of desert ungulates to a water project in Arizona. Journal of Wildlife Management 59:292-300.

Kuck, L., G. L. Hompland, and E. H. Merrill. 1985. Elk calf response to simulated mine disturbance in southeast Idaho. Journal of Wildlife Management 49:751-757.

Leptich, D. J., D. G. Beck, and D. E. Beaver. 1994. Aircraft-based LORAN-C and GPS accuracy for wildlife research on inland study sites. Wildlife Society Bulletin 22:561-565.

Leslie, D. M., Jr., and C. L. Douglas. 1979. Desert bighorn sheep of the River Mountains, Nevada. Wildlife Monographs 66:1-56.

Leslie, D. M., Jr., and C. L. Douglas. 1980. Human disturbance at water sources of desert bighorn sheep. Wildlife Society Bulletin 8:284-290.

MacArthur, R. A., V. Geist, and R. H. Johnston. 1982. Cardiac and behavioral responses of mountain sheep to human disturbance. Journal of Wildlife Management 46:351-358.

Molvar, E. M., and R. T. Bowyer. 1994. Costs and benefits of group living in a recently social ungulate: the Alaskan moose. Journal of Mammalogy 75:621-630.

Morgantini, L. E., and B. W. Worbets. 1988. Bighorn sheep use of a gas wellsite during servicing and testing: a case study of impact and mitigation. Proceedings of the Biennial Symposium of the Northern Wild Sheep and Goat Council 6:159-164.

Nicholson, M. C., R. T. Bowyer, and J. G. Kie. 1997. Habitat selection and survival of mule deer: tradeoffs associated with migration. Journal of Mammalogy 78:483-504.

Oehler, M. W. Sr., V. C. Bleich, and R. T. Bowyer. 1996. Error associated with LORAN-C: effects of aircraft altitude and geographic location. Desert Bighorn Council Transactions 40:19-21.

Oehler, M. W. Sr., R. T. Bowyer, and V. C. Bleich. 2003. Home ranges of mountain sheep: effects of precipitation in a desert ecosystem. Mammalia 69:337-354.

Patric, E. F., T. P. Husband, C. G. McKiel, and W. M. Sullivan. 1988. Potential of LORAN-C for wildlife research along coastal landscapes. Journal of Wildlife Management 32:553-557.

Papouchis, C. M., F. J. Singer, and W. B. Sloan. 2001. Responses of desert bighorn sheep to increased human recreation. Journal of Wildlife Management 65:573-582.

Rachlow, J. L., and R. T. Bowyer. 1994. Variability in maternal behavior by Dall's sheep: environmental tracking or adaptive strategy? Journal of Mammalogy 75:328-337.

Rachlow, J. L., and R. T. Bowyer. 1998. Habitat selection by Dall's sheep (*Ovis dalli*): maternal trade-offs. Journal of Zoology (London) 245:457-465.

Risenhoover, K. L., and J. A. Bailey. 1985. Foraging ecology of mountain sheep: implications for habitat management. Journal of Wildlife Management 49:797-804.

Rubin, E. S., W. M. Boyce, and V. C. Bleich. 2000. Reproductive strategies of desert bighorn sheep. Journal of Mammalogy 81:769-786.

SAS Institute, Inc. 1997. SAS/STAT software changes and enhancements through release 6.12 ed. SAS Institute, Cary, North Carolina, USA.

Solow, A. R. 1989. A randomization test for independence of animal locations. Ecology 70:1546-1549.

Sparks, D. R., and J. C. Malechek. 1968. Estimating percentage dry weight in diets using a microscope technique. Journal of Range Management 21:264-265.

Statistical Package for the Social Sciences. 1993. SPSS for windows: base system user's guide. Release 6.0. SPSS, Chicago, Illinois, USA.

Stewart, K. M., R. T. Bowyer, J. G. Kie, N. J. Cimon, and B. K. Johnson. 2002. Temporospatial distributions of elk, mule deer, and cattle: resource partitioning and competitive displacement. Journal of Mammalogy 83:229-244.

Stockwell, C. A., G. C. Bateman, and J. Berger. 1991. Conflicts in national parks: a case study of helicopters and bighorn sheep time budgets at the Grand Canyon. Biological Conservation 56:317-328.

Tilley, J. M., and R. A. Terry. 1963. A two-stage technique for the *in vitro* digestion of forage crops. Journal of the British Grassland Society 18:104-111.

Van Dyke, F., and W. C. Klein. 1996. Response of elk to installation of oil wells. Journal of Mammalogy 77:1028-1041.

Wehausen, J. D., and M. C. Hansen. 1988. Plant communities as the nutrient base of mountain sheep populations. Pages 256-268 in Plant biology of eastern California (C. A. Hall, Jr., and V. Doyle-Jones, editors). University of California, White Mountain Research Station, Bishop, California, USA.

Welles, R. E., and F. B. Welles. 1961. The bighorn of Death Valley. National Park Service, Washington, D.C., USA.

White, R. G. 1983. Foraging patterns and their multiplier effects on productivity of northern ungulates. Oikos 40:377-384.

Worton, B. J. 1989. Kernel methods for estimating the utilization distribution in home-range studies. Ecology 70:164-168.

Zar, J. H. 1984. Biostatistical analysis. Second edition. Prentice-Hall, Englewood Cliffs, New Jersey, USA.

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DOMOIC ACID IN THE SANTA CRUZ WHARF FISHERY

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ABSTRACT

Fish caught from piers are an important part of the recreational fishery and are not routinely monitored for phycotoxins. Species taken in this catch are often different from those obtained in the commercial catch further offshore, and diets of pier-caught fish may differ from those of offshore species, resulting in different toxin exposure. Here we report preliminary results from a study on diatom toxins in the recreational fish catch, using data collected from 4 July 2001 to 15 December 2001 at the Santa Cruz Municipal Wharf, a heavily fished coastal pier in California. A variety of frequently caught fish species were taken by hook-and-line from the Santa Cruz Wharf and tested for the diatom toxin domoic acid (DA). Fish viscera and muscle tissue were analyzed separately using High Performance Liquid Chromatography methods. DA was detected in fish viscera but not in fish muscle tissue. DA in fish viscera coincided with the appearance of elevated cell counts of species of the DA-producing diatom, *Pseudo-nitzschia australis*, and with DA in the water column. DA was detected almost exclusively in white croaker, *Genyonemus lineatus*, and to a lesser extent in staghorn sculpin, *Leptocottus armatus*. Although the DA levels in these two fish species were comparatively low (<3 µg/g tissue), the bloom of toxic species was comparatively modest at the time. Given the much higher toxin levels encountered in the region at other times, it is likely that these two species could achieve much larger toxin loads during the more intensive blooms of toxic *Pseudo-nitzschia* that occur most years in Monterey Bay. The relationship between toxin levels in the water column and in fish tissue is discussed, as well as the relationship between food habits and the accumulation of domoic acid in white croaker.

INTRODUCTION

The occurrence of harmful algal blooms (HABs) in the coastal marine environment has been shown to pose a serious threat to marine organisms, fisheries, and public

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health. Of particular concern in California's Monterey Bay are HABs caused by diatoms of the genus *Pseudo-nitzschia*. Some species of *Pseudo-nitzschia* in the Bay can produce the potent neurotoxin domoic acid (DA). DA is harmful to a variety of marine organisms as well as humans and is the causative agent of amnesic shellfish poisoning (ASP) in humans. DA poisoning has no known antidote and its symptoms include vomiting, seizures, disorientation, and death (Truelove and Iverson 1994). ASP was first documented in Canada in 1987 when 3 people died and over 100 others suffered adverse, short-term neurological effects after eating cultured blue mussels, *Mytilus edulis*, contaminated with DA produced by *Pseudo-nitzschia* (Perl et al. 1990). This incident led to the current regulatory limit of 20 µg/g DA in seafood, the level at or above which the flesh is considered unsafe for human consumption by the Food and Drug Administration (FDA). In 1991, DA from a toxic bloom of *Pseudo-nitzschia* was responsible for the deaths of over a 100 seabirds in Monterey Bay (Work et al. 1993). In 1998, 70 California sea lions, *Zalophus californianus*, exhibited characteristic symptoms, with more than half of these later dying as a result of DA poisoning (Lefebvre et al. 1999). DA has also been detected in commercially important marine organisms and has been the cause of several fishery closures in California due to contamination of seafood products (G. Langlois, California Department of Health Services, personal communication).

To protect the fishing industry and consumers of the catch, DA monitoring has focused understandably on commercial species. Now that DA has been identified as a public health risk, species such as anchovy, *Engraulis mordax*, and sardine, *Sardinops caeruleus*, are sometimes monitored for toxins in California, especially during blooms of *Pseudo-nitzschia* (G. Langlois, personal communication). However, little is known about the effects of DA on fish taken in the recreational or subsistence fisheries and on the associated public health hazards. Fish caught from piers are an important part of recreational and subsistence fisheries and are mostly unmonitored for phycotoxins.

Relatively little is known about the pier fishing community. Fishing from a public pier or wharf requires no permit or license in California, and thus can not be readily tracked by the usual method used to quantify recreational fishers (i.e. the licensing process). Those who fish from piers and other "man-made structures", however, represent about 11% of the total marine recreational fishers in California (estimate from RecFIN data)[2]. The absolute numbers of marine recreational fishers in the state is unknown, but their catch is often considered to be highly significant, both relative to commercial operations and because of their economic impact on regional communities (Coleman et al. 2004). Species taken in a pier fishery are often different from those targeted in the offshore recreational catch and in the commercial fishery, and species in the pier catch frequently may be species of lower value, as suggested by market price when those same species are commercially available in markets (RecFIN[2], Starr et al.

[2]RecFIN (Recreational Fisheries Information Network). Pacific States Marine Recreational Fisheries Monitoring Database (data on recreational fish catches in California are available up to December, 2003). <http://www.psmfc.org/recfin/>

2002). Thus species in the pier catch have not been the focus of much attention. Additionally, the diets of nearshore fish may lead to toxin exposure that is different from that of species taken offshore, because of differences in prey availability in the nearshore subtidal zone as compared with prey availability offshore. The fishing community that uses piers also may have consumption patterns different from other recreational fishers. For example, the majority of pier fishers in California are represented by a variety of ethnic groups who consume up to 90% of their catch and thus are subsistence fishers; in fact, those of Asian and Latino descent consume three times the recommended amount of fish (PEHAB 2003)[3]. Furthermore, among certain ethnic groups of pier fishers, it is a common practice to consume the whole fish, including viscera. This practice may further increase the possibility of DA exposure, since the toxin tends to be concentrated mostly in viscera (Lefebvre et al. 2002).

Routes by which algal toxins move through marine food webs are still relatively poorly understood. Planktivorous fish such as sardine and anchovy, probably the best known DA vectors, can feed directly on planktonic diatoms and tend to be contaminated with DA only during *Pseudo-nitzschia* blooms, as expected of a water soluble toxin (Lefebvre et al. 2002). These planktivorous fish have sufficiently fine gill rakers to directly filter toxic *Pseudo-nitzschia* from the water and thus can contain very high levels in their viscera during toxic *Pseudo-nitzschia* blooms: values greater than 1800 µg/g have been detected in anchovy in Monterey Bay (Lefebvre et al. 2002). Nearshore, shallow-water fish such as those caught from piers can utilize a wide variety of prey, not only pelagic but also benthic species. Such prey, for example, could include invertebrates such as mussels, clams, and sand crabs, which can directly filter out toxic plankton and have been shown to have detectable DA loads (Perl et al. 1990, Wekell et al. 1994, Ferdin et al. 2002). Additional fish prey may include invertebrate detritivores and deposit-feeders like polychaete worms, which would become contaminated with DA when they consume DA-rich flocculate matter that settles from surface *Pseudo-nitzschia* blooms to depth (Buck and Chavez 1994). Additionally some benthic fish may directly consume contaminated pelagic detritus when it reaches the sea floor. If routes exist for DA to enter the food webs of fish caught in pier-based fisheries, then pier fishers may also be at risk.

As researchers attempt to better understand HAB effects, it has become necessary to study how DA moves through the food web from diatoms to higher trophic level organisms. This study will examine the potential exposure to DA of pier fishers and whether consumption of pier-caught fish poses a public health threat. The objective of this study was to determine whether DA toxins are present in the catch of fishes taken recreationally from piers in Monterey Bay and to ascertain the relative roles of different trophic pathways by which DA may transit in shallow marine food webs. The Santa Cruz wharf was selected as a study site because it is the second most heavily fished ocean pier in California (RecFIN[2], data for 2001-2003) and it was the only California pier where water samples were routinely obtained for quantitative

[3]PEHAB (Public and Environmental Health Advisory Board). Consumption of Contaminated Fish. Sept. 2003. <http://www.cchealth.org/prevention/coalitions/pehab/fishReport.pdf>.

measurements of toxic algal abundance and DA water concentrations during the study period (see below). For this study, data were collected during summer and fall months, a time at which DA blooms commonly occur in central California (Kudela et al., submitted). Samples of fish and water were taken from this site over a 6-month period and analyzed for the presence of DA using High Performance Liquid Chromatography (HPLC-UV).

METHODS

Sample Collection

A sample set of fish was obtained from California's Monterey Bay at the Santa Cruz municipal wharf from 4 July 2001 to 15 December 2001, a period that included a relatively modest toxic *Pseudo-nitzschia australis* event. Fish were sampled 1-2 times per week during the study interval, yielding 1-10 individuals per sampling event. All fish were caught from 0600 hours to 1000 hours using common hook and line methods, the typical local fishing technique. Specimens were pooled for analysis from the seaward end (8-10 m water depth) and the middle of the pier (3-5 m water depth), places where local fishers typically position themselves. Upon capture, the standard length (SL) of each specimen was taken and each fish was identified to species level. The fish were then immediately stored in ice until transferred to a -20°C freezer where they were stored until dissected.

In order to determine the local availability of DA sources for fish during the study interval, water samples were also collected during the July 2001 to 15 December 2001 period. Sea water (SW) samples were taken on a weekly basis from the Santa Cruz wharf for DA analysis by HPLC. SW samples were collected from the surface at the seaward end of the pier using a bucket. Sample water was filtered after collection through a glass fiber filter (Gelmon GF/F) using low vacuum pressure. Five hundred ml of water was filtered from each sample, and was kept frozen at -20°C until extraction for DA. Counts of the toxic cells were also obtained from the same water samples, as described in Lefebvre et al. (2002).

Preparation Of Viscera And Muscle Tissue Extracts

DA from viscera and muscle tissue samples was extracted following procedures described in Lefebvre et al. (1999) and Hatfield et al. (1994). DA was analyzed from whole viscera (including all organs in the intracoelomic cavity) for all fish taken. DA was analyzed in muscle tissue only in specimens in which visceral DA was detected. Samples were prepared and analyzed within 3 weeks. In order to minimize post-mortem leakage of DA from the digestive tract, fish were kept frozen until processed and whole viscera were removed from fish without thawing. Muscle tissue samples were taken from the posterior dorsal side of the fish near the tail to avoid regions where DA may have diffused from the digestive tract (Lefebvre et al. 2001). Aliquots taken from the same species of fish on the same day were pooled before extraction. Whole viscera were

removed from fish and homogenized with mortar and pestle. A 4-g subsample of the homogenate was added to 16 ml of 50% methanol and homogenized for 2 minutes at highest setting with homogenizer probe. The homogenate was then sonicated for 2 minutes in a sonicator bath, followed by centrifugation for 20 minutes at 4000 × g. The supernatant was then vacuum-filtered through a 1.0 µm polycarbonate filter, in preparation for solid-phase extraction.

Solid-Phase Extraction of Viscera And Muscle Tissue

Solid-phase extraction was performed on all samples using JT Baker strong anion exchange (SAX) cartridges. SAX columns were first conditioned with 6 ml of Nanopure water, 3 ml of 100% MeOH, and 3 ml of 50% MeOH. Two ml of sample filtrate was passed through the column, followed by 5 ml of wash solution (1:9 MeCN:H_2O). DA in the column was eluted with 5 ml of eluting solution (0.5M NaCl in 10% MeCN), then analyzed by HPLC. Efficiency of extraction was 83.6 ± 5.7% (n = 3) for whole viscera and 78.2 ± 11.3% for muscle tissue. Method detection limits were 1.5 ppm for viscera and 1.6 ppm for muscle tissue.

Water Sample Preparation

DA was extracted and derivatized from water samples using procedures outlined in Wright and Quilliam (1995). Filters were homogenized in 10% methanol and ground using a Teflon pestle at 250 rpm, followed by sonication for 3 minutes. Samples were then centrifuged for 1 minute at 4000 rpm and supernatant was filtered through a 0.22 µm filter.

Two hundred µl of filtered sample extract was added to 50 µl of borate buffer, 10 µl of dihydrokainic acid (DHK) internal standard, and 250 µl of 9-Fluorenylmethoxy-chloroformate (FMOC-Cl) reagent solution, then vortexed for 45 seconds. The sample was then washed three times with 500 µl of ethyl acetate to remove excess FMOC-Cl. Derivatized samples were run on HPLC for DA analysis.

High-Performance Liquid Chromatography

DA from samples of whole viscera and muscle tissue was measured using a Hewlett-Packard 1090 HPLC instrument. The HPLC was equipped with a diode array detector set at 242 nm with a bandwidth of 10 nanometers (nm). Analysis was done using an isocratic elution profile. A reference signal was set at 450 nm with a bandwidth of 10 nm. The HPLC was equipped with a Vydac reverse-phase C18 column (catalog #201TP52, 21 × 25 mm; Separations Group, Hesperia, Calif.) and a Vydac guard column (particle size 5 µm). The mobile phase used was 90% H_2O, 10% MeCN, 0.1% TFA and was run at a flow rate of 0.3 ml/min. Prior to analysis, the solvent was degassed with helium for 10 minutes. All injections were 20 µl each. Samples had a 15-minute run time.

Using DACS-1C domoic acid standard, a calibration curve was generated using standards of 0.15, 0.3, 0.5, 1.0, 2.0, 4.0, 8.0, and 16.0 ppm (r = 0.99). The lowest detectable

standard was 0.15 ppm. The instrument limit of detection (the concentration corresponding to three times the SD of the signal from the lowest detectable standard) was 0.1 ppm. Spike and recovery experiments using DA standards yielded 83 ± 4.7% and 78 ± 11.3% efficiency for viscera and muscle tissue samples, respectively.

SW samples were analyzed for DA using a Hewlett-Packard 1090 HPLC, equipped with a Hewlett-Packard 1046A programmable fluorescence detector (FLD). The FLD was set at 264 nm excitation and 313 nm emission with a photomultiplier gain of 13. The analysis was done using an isocratic elution profile. The HPLC was equipped with a Vydac reverse-phase C18 column and a Vydac guard column (particle size 5 µm). The mobile phase used was 60% H_2O, 40% MeCN, 0.1% TFA and was run at a flow rate of 0.2 ml/min. Prior to analysis, the solvent was degassed with helium for 10 minutes. All injections were 5 µl each. Samples had a 30-minute run time. FMOC-Cl was used as a fluorescent tag for DA analysis of all water samples.

Using DACS-1C domoic acid standard, a calibration curve was generated using standards of 5, 10, 25, 50, 100, and 250 nanograms (ng)/ml ($r = 0.99$). The lowest detectable standard was 5 ng/ml. The instrument limit of detection was 1.4 ng/ml. Spike and recovery experiments using DA standards yielded 84 ± 7% efficiency.

Chemical Reagents

Reagents used for calibration standards and spike/recovery calculations included DACS-1C certified DA standard (National Research Council of Canada, Institute for Marine Biosciences, 1411 Oxford Street, Halifax, NS, Canada), and Sigma DA reagent (90% pure). Standards were stored under refrigeration in the dark. Optima grade methanol, acetonitrile, ethyl acetate, analytical grade NaCl, and trifluoroacetic acid were obtained from Fisher Scientific (Pittsburgh, PA). DHK was used as an internal standard and FMOC-Cl was used as a fluorescing reagent for detection of amino acids (Aldrich Chemical Co. Milwaukee, WI). Milli-Q water was used in preparation of solutions.

RESULTS

The sample set consisted of 115 specimens, representing 12 different species of fish. The most abundant species caught was white croaker, *Genyonemus lineatus*, accounting for 23% of the total catch during the sampling period (Table 1). Also abundant were juvenile and adult shiner surfperch, *Cymatogaster aggregata* (22%), juvenile bocaccio, *Sebastes paucispinis* (21%), and juvenile and adult staghorn sculpin, *Leptocottus armatus* (14%). These fish were common in the catch of most fishers at the wharf at the time. Toward the end of the sampling period, the catch consisted mainly of white croaker and staghorn sculpin.

HPLC analysis of whole viscera detected DA contamination in white croaker on four separate catch dates and in staghorn sculpin on one date (Table 2). DA levels in white croaker ranged from 2.0 to 2.8 µg/g tissue. The DA level detected in staghorn sculpin was 2.1 µg/g. DA was not detected in the muscle tissue of either fish. All other species of fish in the sample set showed no detectable amounts of DA (Table 2). (While the

Table 1. Frequency of occurrence of fish captured at Santa Cruz Pier (n = 120), 4 July to 15 December 2001.

	% of sample set
White croaker, *Genyonemus lineatus*	23
Shiner surfperch, *Cymatogaster aggregata*	22
Bocaccio, *Sebastes paucispinis*	21
Staghorn sculpin, *Leptocottus armatus*	14
Pacific sardine, *Sardinops caeruleus*	11
Walleye surfperch, *Hyperprosopon argenteum*	5
Striped surfperch, *Embiotoca lateralis*	< 1
Silver surfperch, *Hyperprosopon ellipticum*	< 1
Pile perch, *Rhacochilus vacca*	< 1
Pacific mackerel, *Scomber japonicus*	< 1
Jacksmelt, *Atherinopsis californiensis*	< 1
Black perch, *Embiotoca jacksoni*	< 1

Table 2. Distribution of DA detected in fish muscle tissue and viscera

	Date	Species	*n*	SL (cm)	Viscera DA (µg/g)	Muscle DA (µg/g)
Pre-	7/4/2001	white croaker	1	14.0	0	
Bloom	7/4/2001	bocaccio	1	12.0	0	
	7/4/2001	black perch	1	14.5	0	
	7/13/2001	bocaccio	2	10.0-12.0	0	
	7/13/2001	walleye surfperch	4	12.0-13.0	0	
	7/21/2001	staghorn sculpin	1	16.5	0	
	7/21/2001	shiner surfperch	4	8.0-8.5	0	
	7/21/2001	pile perch	1	17.5	0	
	8/3/2001	pacific sardine	5	18.0-19.0	0	
	8/10/2001	bocaccio	1	14.5	0	
	8/11/2001	pacific sardine	3	19.0-21.0	0	
	8/11/2001	jacksmelt	1	17.0	0	
	8/18/2001	pacific sardine	5	18.0-23.0	0	
	8/18/2001	pacific mackerel	1	24.0	0	
	8/21/2001	white croaker	2	11.0-17.0	2.4	0
	8/25/2001	shiner surfperch	4	9.0-12.0	0	
	8/25/2001	walleye surfperch	1	17.5	0	
	8/25/2001	striped surfperch	1	14.0	0	
	8/31/2001	shiner surfperch	6	9.0-9.5	0	
	9/10/2001	white croaker	2	11.0-16.0	0	
	9/10/2001	bocaccio	8	10.0-13.0	0	
	9/13/2001	shiner surfperch	6	9.0-10.5	0	
	9/19/2001	shiner surfperch	4	10.0-11.0	0	

Table 2 (continued)

	Date	Species	*n*	SL (cm)	Viscera DA (µg/g)	Muscle DA (µg/g)
	9/19/2001	bocaccio	5	9.0-10.0	0	
	9/24/2001	staghorn sculpin	4	13.5-14	0	
	9/28/2001	staghorn sculpin	1	12.0	0	
	10/2/2001	white croaker	2	13.0-20.0	0	
During	10/10/2001	white croaker	6	12.0-16.0	2.8	0
Bloom	10/17/2001	shiner surfperch	2	10.0-11.0	0	
	10/17/2001	bocaccio	5	10.0-11.0	0	
	10/17/2001	walleye surfperch	1	16.0	0	
	10/23/2001	staghorn sculpin	1	12.0	0	
	10/23/2001	silver surfperch	1	15.0	0	
	10/26/2001	white croaker	1	6.0	0	
	10/26/2001	staghorn sculpin	1	16.0	0	
Post-	10/29/2001	white croaker	3	12.0-16.0	2.4	0
Bloom	10/29/2001	staghorn sculpin	3	11.0-14.0	0	
	11/2/2001	white croaker	3	10.0-16.5	2	0
	11/2/2001	staghorn sculpin	1	12.5	2.1	0
	11/2/2001	bocaccio	3	10.5-11.0	0	
	11/9/2001	staghorn sculpin	1	13.0	0	
	11/13/2001	white croaker	2	16.0-18.0	0	
	11/17/2001	white croaker	2	17.0	0	
	11/17/2001	staghorn sculpin	2	15.0	0	
	11/21/2001	white croaker	1	15.0	0	
	11/21/2001	staghorn sculpin	2	12.0-15.0	0	
	12/4/2001	white croaker	1	18.0	0	
	12/15/2001	white croaker	1	15.0	0	

HPLC method used in this study is not as sensitive as other methods used to detect DA, it can be used to reliably detect tissue burdens ≥ 1.5 µg/g, well below the regulatory limit for human consumption. What is not known at present is the effects of chronic low level exposures to DA).

The collection period coincided with a toxic algal, *Pseudo-nitzschia australis*, bloom in Monterey Bay. Cell counts for this bloom reached 1.6×10^4 cells/L at the Santa Cruz wharf. DA was detected in SW samples during this period reaching concentrations of 339 ng/l at the Santa Cruz wharf.

DISCUSSION

The contamination of pelagic food webs by DA has been well-established in Monterey Bay. The northern anchovy appears to be a particularly important vector of the toxin (Lefebvre et al. 2002). In May 1998, a large-scale California sea lion stranding

event along the central California coast resulted in the deaths of 47 animals and coincided with a toxic bloom of *P. australis* in the region (Gulland 1999[4], Scholin et al. 2000). DA was later identified as the probable causative agent in this event (Lefebvre et al. 1999), vectored by the anchovies. During the course of this study, 16 sea lions were found stranded along the central California coast (two of these in Santa Cruz County), and these were confirmed to be contaminated with DA (F. Gulland, Director of Veterinary Science, The Marine Mammal Center, personal communication). Commercially-caught sardines landed in the bay during this period and analyzed by the California Department of Health Services, contained DA at concentrations reaching 51 µg/g (Langlois 2002)[5]. However, most of the fish taken in this pier study were not pelagic, but benthic-feeding fish. Much less is known about the DA contamination of benthic food webs than of pelagic food webs and thus about the potential risk to the human population of consuming bottom fish.

The DA detected in white croaker (2.0-2.8 µg/g) and in staghorn sculpin (2.1 µg/g) was well below the FDA regulatory limit of 20 µg/g. However, DA levels detected in Santa Cruz water during the associated *Pseudo-nitzschia* event were modest in comparison to levels detected in the region during typical blooms (e.g., Walz et al. 1994, Scholin et al. 2000). Since the event that occurred during this study was relatively modest, it is reasonable to expect much higher DA levels in fish during major blooms. Furthermore, DA values detected in water at other Monterey Bay sites during the same bloom are typically much higher than those detected at the Santa Cruz wharf (M. Silver, unpublished), possibly explaining the higher DA concentrations found in sardines from the commercial catch. Indeed, for the bloom described in this study, DA levels in Santa Cruz water were an order of magnitude lower than those detected at a more southerly site in the bay (S. Fire and M. Silver, unpublished). It is also important to note that the HPLC-UV detection limits here are rather high and that DA may in fact have been present in low concentrations in other fish analyzed in this study.

An intriguing aspect of this study is the presence of DA in white croaker and staghorn sculpin when it was not measurable in other species during the toxic *Pseudo-nitzschia* event. Although several other species of fish were caught during times when DA was found in water samples, they did not have detectable levels of DA. During the 3-week bloom period there were six species of fish sampled, and only white croaker contained measurable DA levels (Table 2). Four out of the five samples contaminated with DA during the study period were from white croaker, suggesting that this species is somewhat unique among pier fish in its ability to accumulate the toxin. Such results are not totally unexpected, however, as this species is known to accumulate a variety of toxins available in other nearshore settings (e.g. Greenfield et al. 2003)

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[4]Gulland, F. 1999. Domoic acid toxicity in California sea lions stranded along the central California coast, May-October 1998. NOAA Tech. Memo. NMFS-OPR-8. USA National Marine Fisheries Service, U.S. Department of Commerce.

[5]Langlois, G. W. 2002. Marine Biotoxin Monitoring Program annual report 2001. California Dept. of Health Services, Division of Drinking Water and Environmental Health. Berkeley, CA.

The feeding habits of white croaker may explain its DA contamination. According to Ware (1979)[6], white croaker exhibits "euryphagous [feeding] habits and is capable of utilizing all available food resources" in the area. Foraging individuals apparently suck up sediment in order to ingest prey items (Ware 1979)[6]. This species' broad diet is dominated by epifaunal and infaunal invertebrates. Adult white croaker (over 100 mm length) forage mainly on deposit feeding polychaetes, detrital feeding macrocrustaceans and have even been observed to eat "large amounts of green algae, *Ulva* and *Enteromorpha*, and amorphous flocculent matter" (Ware 1979)[6]. In this study, 96% of white croaker individuals sampled were over 100 mm SL and thus were considered adults. It seems plausible that this euryphagous feeding behavior may be responsible for the accumulation of DA in white croaker. The contamination pathway for this species of fish could be either the flocculent detrital matter or the organisms that consume it. Given that other crustacean-feeding fish were sampled at the same time as white croaker and only one date of staghorn sculpin was DA-toxic, this suggests that it may be a detrital pathway and not an invertebrate pathway.

DA-contaminated fish also were caught before, during and immediately following the bloom. White croaker viscera contained toxin on two dates during the week following the bloom, and staghorn sculpin viscera had measurable DA levels on one date during that week (Table 2). During the post-bloom period, *P. australis*, like some other diatoms, can flocculate and settle through the water column (Buck and Chavez 1994) including to shallow, nearshore sediments in Monterey Bay (R. Kvitek, California State University, Monterey Bay and V. Welborn, University of California, Santa Cruz, personal communication). This influx of detrital material becomes available to benthic populations, and may provide a DA source to benthic consumers even after the surface bloom is gone. The feeding habits of staghorn sculpin may be an explanation for its DA contamination, as they are somewhat similar to those of white croaker. As suggested above, benthic invertebrates may be the mechanism of DA transfer in this case as well. According to Barry et al. (1996), adults of staghorn sculpin consume mainly "small shore crabs, gammarid amphipods ... and macroalgae". Small individuals fed on smaller, less mobile prey such as gammarid amphipods, harpacticoid copepods, and polychaete worms (Barry et al. 1996). Indeed, the small shore crab, *Emerita analoga*, has been shown to contain high levels of DA during HABs (Ferdin et al. 2002, Goldberg 2003[7]).

Shiner surfperch and bocaccio were nearly as abundant as white croaker, and more abundant than staghorn sculpin in the sample set, but showed no detectable levels of DA. This may have been due to differences in feeding habits and/or due to their take occurring primarily before the bloom began. Catches of shiner surfperch and bocaccio were frequent during the beginning of the sampling period but had become infrequent

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[6]Ware, R. R. 1979. The food habits of white croaker *Genyonemus lineatus* and an infaunal analysis near areas of waste discharge in outer Los Angeles Harbor. M.A. Thesis, California State Univ., Long Beach, 113 p.

[7]Goldberg, J.D. 2003. Domoic acid in the benthic food web of Monterey Bay, California. Master's Thesis, California State University Monterey Bay, CA. 44 p.

by the time the bloom peaked. Moreover, even if present, juvenile bocaccio of the size collected are already piscivorous and no longer feed on benthic invertebrates (M. Carr, Professor, University of California at Santa Cruz, personal communication). Shiner surfperch feeds mainly on epifaunal worms (*Armandia brevis* and other polychaetes) and crustaceans such as harpacticoid copepods and gammarid amphipods (Barry et al. 1996). The four most abundant fish in this study have several prey items in common, but there may be some other mechanism at work to bring DA into contact with white croaker, such as the consumption of sediment as part of its feeding behavior. It is possible that the ingestion of flocculated algal cells and DA-contaminated detritus during feeding could make this species more likely to become toxic. The high abundance of white croaker in the catch relative to other species during the bloom period may be another explanation for it exhibiting most of the DA contamination.

The relationship between the appearance of DA in croaker with that of DA and toxic *Pseudo-nitzschia* in the water is not yet clear from the results of this study, given the relatively short duration of the project and the low levels of toxin in the water and fish viscera. A surprising result of this preliminary study, however, was the detection of DA in white croaker on 21 August, almost 2 months before the bloom began. If such a result is found again, the mismatch could indicate the presence of DA reservoirs in sediments or in prey items during non-bloom times at concentrations high enough to contaminate benthic populations. Some organisms, such as sardine and anchovy, are found to be toxic only when DA is present in the water column (Lefebvre et al. 2001). However, there is evidence that some other organisms can retain DA long after bloom conditions subside: razor clams are toxic up to 86 days after removal from their natural environment to the laboratory (Horner et al. 1993, Wekell et al. 1994). The mechanism by which this water soluble toxin is retained is not yet understood. Very recently, Goldberg (2003)[7] found a benthic echiurid worm, *Urechis caupo*, to be toxic whenever sampled in the bay, suggesting some unusually effective method for retaining the toxin or, alternately, access to environmental reservoirs of the toxin heretofore undetected. Similar retention of DA in prey organisms of white croaker could explain DA contamination during periods when DA is not detectable in the water column.

The levels of DA found in white croaker and staghorn sculpin viscera were below 20 µg/g (Tables 2), the level considered unsafe for human consumption. However, the fact that white croaker contained DA on four dates during the sampling period suggests the possibility of a potential hazard to the recreational and subsistence fishery. Although most fishermen hold little regard for this species, it is popular with certain ethnic groups, especially in the Asian community (Love et al. 1984). In fact, fish cake plants that processed up to 4000 lb of white croaker per day sold their product exclusively to Asian communities in Los Angeles (Love et al. 1984). During the sampling period for this study, fishers of Asian descent were observed to take and keep them in their catch and expressed their intent to consume them (S. Fire, personal observation). Today, white croaker is still fished all over Monterey Bay throughout the year and represents 4% of the catch at the Santa Cruz Wharf in the last 3 years for which data are available (RecFIN[2], data for 2001-2003) During this same period, this species represented ~7% of the pier catch in the state (RecFIN)[2].

White croaker did not always contain DA during the fall 2001 bloom, however, and when it did, it was found only in the viscera. Although DA was not detected in the muscle tissue of any fish, it is not an uncommon practice to consume whole fish, including viscera. This practice could represent the greatest hazard to fishers. Clearly, it is advisable to remove all viscera from white croaker before consumption, even when a harmful algal bloom is not in evidence.

In light of these data, public health may be a concern when involving the pier fisher community. The Environmental Protection Agency states that subsistence fishers rely on noncommercial fish as a major source of protein, and do so for a greater portion of the year than other fishers (US EPA 1994)[8]. Studies have found that fish consumption rates were higher for Asians, African-Americans, Native Americans, and other ethnic groups and the parts of the fish eaten by these groups varied as well (OEHHA 2001)[9]. In the event that fish such as white croaker become contaminated with DA during a moderate to severe algal bloom, certain groups of pier fishers may be consuming levels of the toxin that may approach or exceed FDA levels. Furthermore, these fishers who already may be at risk may be exposed to this risk for longer periods of time in comparison to other types of fishers.

CONCLUSIONS

The presence of DA in the viscera of commonly caught pier fish establishes a potential public health risk to pier fishers. Some of these fish contain detectable levels of phycotoxins even during low-level algal blooms, suggesting the possibility for higher levels of intoxication during severe blooms. Individuals who fish for sport or as a means to provide food for their families may put themselves at risk for this toxin when taking fish close to toxic bloom events, especially if their catch includes white croaker and staghorn sculpin. This risk may be compounded by the practice of consuming fish that have not been gutted, essentially when eating the organs that concentrate toxins. Until more is known about the trophic transfer of DA from phytoplankton to fish, and about which species of nearshore fish are susceptible to DA-contamination, the subsistence and recreational fisheries should be alerted to the potential health risk of consuming these species.

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[8]U.S. EPA (1994). *Guidance for Assessing Chemical Contaminant Data for Use in Fish Advisories, Vol. II: Risk Assessment and Fish Consumption Limits.* Office of Water. U.S. Environmental Protection Agency. Washington, DC: Document No. EPA 823-B-94-004.

[9]OEHHA (Office of Environmental Health Hazard Assessment). Chemicals in fish: Consumption of fish and shellfish in California and the United States. Oct. 2001. <http://www.oehha.org/fish/pdf/Fishconsumptionrpt.pdf>.

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LITERATURE CITED

Barry, J. P., M. M. Yoklavich, G. M. Cailliet, D. A. Ambrose, and B. S. Antrim. 1996. Trophic ecology of the dominant fishes in Elkhorn Slough, California, 1974-1980. Estuaries 19(1):115-134.

Buck, K. R., and F. P. Chavez. 1994. Diatom aggregates from the open ocean. Journal of Plankton Research 16(10):1449-1457.

Coleman, F. C., W. F. Figueira, J. S. Ueland, and L. B. Crowder. 2004. The impact of United States recreational fisheries on marine fish populations. Science 305:1958-1960.

Ferdin, M. E., R. G. Kvitek, C. K. Bretz, C. L. Powell, G. J. Doucette, K. A. Lefebvre, S. Coale, and M. W. Silver. 2002. *Emerita analoga* (Stimpson) – possible new indicator species for the phycotoxin domoic acid in California coastal waters. Toxicon 40:1259-1265.

Greenfield, B.K., J. A. Davis, R. Fairey, C. Roberts, D. B. Crane. G. Ichikawa, and M. Petreas. 2003. Contaminant concentrations in fish from San Francisco Bay, 2000. RMP Technical Report: SFEI Contribution 77, San Francisco Estuary Institute, Oakland, CA.

Hatfield, C. L., J. C. Wekell, E. J. Gauglitz Jr., and H. J. Barnett. 1994. Salt cleanup procedure for the determination of domoic acid by HPLC. Natural Toxins 2:206-211.

Horner, R. A., M. B. Kusske, B. P. Moynihan, R. N. Skinner, and J. C. Wekell. 1993. Retention of Domoic Acid by Pacific Razor Clams, *Siliqua patula* (Dixon, 1789): Preliminary study. Journal of Shellfish Research 12(2):451-456.

Kudela, R., W. Cochland, and A. Roberts. 2003. Spatial and temporal patterns of pseudo-nitzschia spp. in central California related regional oceanography. In: Steidinger, K. A., J. H. Landsberg, C. R. Tomas and G. A. Vargo (Eds) Harmful Algae 2002. Florida and Wildlife Conservation Commission, Florida Institute of ceanography, and Intergovernmental Oceanographic Commission of UNESCO, in press.

Lefebvre, K. A., C. L. Powell, M. Busman, G. J. Doucette, P. D. R. Moeller, J. B. Silver, P. E. Miller, M. P. Hughes, S. Singaram, M. W. Silver, and R. S. Tjeerdema. 1999. Detection of Domoic Acid in Northern Anchovies and California Sea Lions Associated with an Unusual Mortality Event. Natural Toxins 7:85-92.

Lefebvre, K. A., S. L. Dovel, and M. W. Silver. 2001. Tissue distribution and neurotoxic effects of domoic acid in a prominent vector species, the northern anchovy *Engraulis mordax*. Marine Biology 138:693-700.

Lefebvre, K. A., M. W. Silver, S. L. Coale, and R. S. Tjeerdema. 2002. Domoic acid in planktivorous fish in relation to toxic *Pseudo-nitzschia* cell densities. Marine Biology 140(3):625-631.

Love, M. S., G. E. McGowen, W. Westphal, R. J. Lavenberg, and L. Martin. 1984. Aspects of the life history and fishery of the white croaker, *Genyonemus lineatus* (Sciaenidae), off California. Fishery Bulletin 82:179-198.

Malins, D. C., B. B. McCain, D. W. Brown, M. S. Myers, M. M. Krahn, and S. L. Chan. 1987. Toxic Chemicals, Including Aromatic and Chlorinated Hydrocarbons and Their Derivatives, and Liver Lesions in White Croaker (*Genyonemus lineatus*) from the vicinity of Los Angeles. Environmental Science & Technology 21(8):765-770.

McCain, B. B., S. L. Chan, M. M. Krahn, D. W. Brown, M. S. Myers, J. T. Landahl, S. Pierce, R. C. Clark Jr., and U. Varanasi. 1992. Chemical Contamination and Associated Fish Diseases in San Diego Bay. Environmental Science & Technology 26(4):725-733.

Perl, T. M., L. Bedard, T. Kosatsky, J. C. Hockin, E. C. D. Todd, and R. S. Remis. 1990. An outbreak of toxic encephalopathy caused by eating mussels contaminated with domoic acid. New England Journal of Medicine 322(25):1775-1780.

Scholin, C. A., F. Gulland, G. J. Doucette, S. Benson, S. Busman, F. P. Chavez, J. Cordaro, R. DeLong, A. DeVogelaere, J. Harvey, M. Haulena, K. Lefebvre, T. Lipscomb, S. Loscutoff, L. J. Lowenstine, R. Marin, P. E. Miller, W. A. McLellan, P. D. R. Moeller, C. L. Powell, T. Rowles, P. Silvagni, M. Silver, T. Spraker, V. Trainer, and F. M. Van Dolah. 2000. Mortality of sea lions along the central California coast linked to toxic diatom blooms. Nature 403:80-84.

Starr, R. M., J. M. Cope and L. A. Kerr. 2002. Trends in fisheries and fishery resources associated with the Monterey Bay National Marine Sanctuary from 1981-2000. California Sea Grant College Program, University of California, Publication #T-046, 156 pp.

Truelove, J., and F. Iverson. 1994. Serum domoic acid clearance and clinical observations in the Cynomolgus monkey and Sprague-Dawley rat following a single IV dose. Bulletin of Environmental Contamination and Toxicology 52:479-486.

Walz, P. M., D. L. Garrison, W. M. Graham, M. A. Cattey, R. S. Tjeerdema, and M. W. Silver. 1994. Domoic acid producing diatom blooms in Monterey Bay, California: 1991-1993. Natural Toxins 2:271-279.

Wekell, J. C., E. J. Gauglitz, H. J. Barnett, C. L. Hatfield, D. Simons, and D. Ayres. 1994. Occurrence of domoic acid in Washington state razor clams (*Siliqua patula*) during 1991-1993. Natural Toxins 2:197-205.

Work, T. M., A. M. Beal, L. Fritz, M. A. Quilliam, M. Silver, K. Buck, and J. L. C. Wright. 1993. Domoic acid intoxications of brown pelicans and cormorants in Santa Cruz, California. Pages 643-649 *In*: Smayda T.J., (editors), Toxic Phytoplankton Blooms in the Sea, Elsevier, Amsterdam, pp..

Wright, J. L. C. and M. A. Quilliam. 1995. Methods for Domoic Acid, the Amnesic Shellfish Poison. Pages 113-133 *In*: Manual on Harmful Marine Microalgae. Intergovernmental Oceanographic Commission Manuals and Guides No. 33, UNESCO, Paris.

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TOTAL MERCURY CONCENTRATIONS IN FILLETS OF BLUEGILL, REDEAR SUNFISH, LARGEMOUTH BASS, AND OTHER FISHES FROM LAKE NATOMA, SACRAMENTO COUNTY, CALIFORNIA

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This study was conducted during September-October 2002 to verify preliminary findings of elevated total mercury concentrations in skinless fillets of sportfishes inhabiting Lake Natoma. Although we measured total mercury concentrations, most mercury in fish flesh occurs in the methylated form. In August 2000, other investigators collected a small number of fish containing mercury concentrations that exceeded 0.30 µg/g wet weight, the U.S. Environmental Protection Agency (USEPA) tissue residue criterion derived from a reference dose for methylmercury that may cause undesirable neurological abnormalities in human infants exposed *in utero* when pregnant women consume mercury-contaminated foods. During our study, skinless fillets of bluegill, *Lepomis macrochirus*, contained as much as 0.19 µg Hg/g wet weight (1.06 µg Hg/g dry weight); redear sunfish, *L. microlophus*, contained as much as 0.39 µg Hg/g wet weight (1.99 µg Hg/g dry weight); and largemouth bass, *Micropterus salmoides*, contained as much as 0.86 µg Hg/g wet weight (3.85 µg Hg/g dry weight). Maximum concentrations of mercury in other fish species varied from 0.097 µg/g wet weight (0.537 µg/g dry weight) in rainbow trout, *Oncorhynchus mykiss*, to 0.56 µg/g wet weight (3.07 µg/g dry weight) in white catfish, *Ameiurus catus*. Altogether, 1 of 20 redear sunfish, 14 of 61 largemouth bass, 1 of 1 brown bullhead, *A. nebulosus*, 2 of 3 spotted bass, *M. punctulatus*, and 1 of 1 white catfish exceeded the USEPA fish tissue methylmercury residue criterion. Only bluegill and largemouth bass exhibited significant correlations between fish total length (TL), weight, and age, and total

mercury concentration in fillets. Judging from a best-fit power-curve equation, largemouth bass measuring 273 mm TL (roughly 292 g) or larger are estimated to contain total mercury concentrations in their fillets that exceed the USEPA fish tissue methylmercury criterion. These results confirmed that some fish species inhabiting Lake Natoma are contaminated with undesirably high concentrations of mercury in their skinless fillets.

INTRODUCTION

Mercury contamination from historic gold mining operations is widespread in many rivers, lakes, and reservoirs on the western slopes of the Sierra Nevada (Alpers and Hunerlach[1] 2000). Miners used mercury to recover gold from placer (alluvial) mines, which used hydraulic, drift, and dredging methods, and from hardrock (lode) mines. At hydraulic mining operations, placer ores were eroded with water cannons and the resulting slurry was directed through sluices and drainage tunnels where the gold particles were combined with liquid mercury to form gold-mercury amalgam. Bowie (1905) estimated that 10-30% of the mercury used in this process was lost each season, resulting in highly contaminated sediments downstream from the mines. According to Alpers and Hunerlach[1] (2000), the annual loss of mercury from a typical sluice was likely several hundred kilograms during the operating season or roughly 1.4-3.6 million kg statewide since 1850. Although mercury was also used in drift mining operations, in dredging operations, and at hardrock mines, the amounts lost to the environment by these sources have not been estimated.

Several ongoing studies by the U.S. Geological Survey (USGS) are assessing the status and dynamics of mercury contamination associated with past hydraulic mining operations in the Sierra Nevada (http://ca.water.usgs.gov/mercury/). By comparison, little work is being conducted on mercury contamination in historically dredged environments in urbanized areas (e.g., the cities of Folsom, Gold River, and Rancho Cordova) within the American River watershed. Lake Natoma, a 162-ha reservoir on the American River, is managed by the U.S. Bureau of Reclamation for multiple uses that include fishing and other water-based recreation. Preliminary results from 22 fish samples collected in Lake Natoma during August 2000 indicated that total mercury concentrations in skinless fillets may approach or exceed guidelines for human consumption (Bowman[2] 2002). One guideline established by the U.S. Environmental Protection Agency (USEPA) is 0.3 µg Hg/g wet weight, the fish tissue residue criterion derived from a reference dose for methylmercury that "...should not be exceeded to protect the health of consumers of noncommercial freshwater/estuarine fish" (USEPA[3]

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[1]Alpers, C.N., and M.P. Hunerlach. 2000. Mercury contamination from historic gold mining in California. U.S. Geological Survey Fact Sheet FS-061-00, Sacramento, California.

[2]Bowman, C. 2002. Mercury taints Natoma fish; runoff from Folsom may be increasing contamination levels. The Sacramento Bee, Sacramento, California (http://www.sacbee.com/content/news/v-print/story/1507238p-1583930c.html).

[3]USEPA (U.S. Environmental Protection Agency). 2001. Water quality criterion for the protection of human health: methyl mercury. EPA-823-R-01-001, Washington, DC. (http://www.epa.gov/waterscience/criteria/methylmercury/).

2001). Due to the relatively high mercury concentrations measured in preliminary samples, we conducted a more intensive follow-up study to better understand the magnitude of mercury contamination in muscle tissue or fillets of sport fishes from Lake Natoma.

Our study was part of a broader investigation of mercury dynamics and bioaccumulation in urbanized areas within the American River watershed that were historically subjected to gold dredging activities. Specific objectives of the study were as follows: (i) to determine if total mercury concentrations in fillets varied according to size, age, or body condition of fishes, (ii) to determine if total mercury concentrations in fillets varied spatially among selected fish species, and (iii) to determine if total mercury concentrations in fillets approach or exceed criteria for human health concerns.

STUDY AREA AND METHODS

Fish were collected from Lake Natoma on the American River between 17 September and 15 October 2002 at five general locations as follows (roughly from upstream to downstream): the Negro Bar vicinity, the mouth of Willow Creek, the Mississippi Bar vicinity, the mouth of Alder Creek, and adjacent to Nimbus Dam (Fig. 1). Lake Natoma is located about 18.6 km east of Sacramento, near the City of Folsom. Water in Lake Natoma originates primarily from upstream releases at Folsom Dam, with lesser inflows from Willow and Alder creeks and other sources. Surface-water characteristics at the combined sampling sites during this time period varied as follows: temperature, 16.0-20.7°C; dissolved oxygen, 7.2-10.3 mg/l; pH, 6.7-7.4; specific conductance, 0.03-0.07 µmhos/cm @ 25°C; and turbidity, 0.0-0.8 nephelometric turbidity units.

Although bluegill, *Lepomis macrochirus*, and largemouth bass, *Micropterus salmoides*, were the primary species targeted for capture with a boat-mounted electroshocker, we also retained small numbers of green sunfish, *L. cyanellus*, redear sunfish, *L. microlophus*, smallmouth bass, *M. dolomieu*, spotted bass, *M. punctulatus*, white catfish, *Ameiurus catus*, black bullhead, *A. melas*, brown bullhead, *A. nebulosus*, and rainbow trout, *Oncorhynchus mykiss*.

We attempted to include both catchable and subcatchable sizes of fish in our samples. Although the designation of a minimum size for catchable fish is often arbitrary, this study adopted either the minimum legal size limit established by the State of California (CDFG[4] 2002) or the minimum sizes for human consumption suggested by various authors. In Lake Natoma, the largemouth bass fishery is regulated by a minimum size restriction of 305 mm, whereas other fishes have no legal size restrictions (CDFG[4] 2002). However, Bennett (1971) indicated that anglers generally keep fish of the following minimum sizes: for bluegill and other sunfishes, 152 mm; and for bullheads (close relatives of the white catfish), 178 mm. According to Leitritz (1970, cited by Edmondson[5] 2002), a "harvestable trout" is a fish between 178 and 254 mm or

[4]CDFG (California Department of Fish and Game). 2002. Freshwater sport fishing regulations booklet. California Fish and Game Commission, Sacramento, California.

[5]Edmondson, J. 2002. Setting priorities—the benefits of revamping the California Department of Fish and Game catchable trout hatchery project. California Trout, Inc., San Francisco, California (Unpublished report).

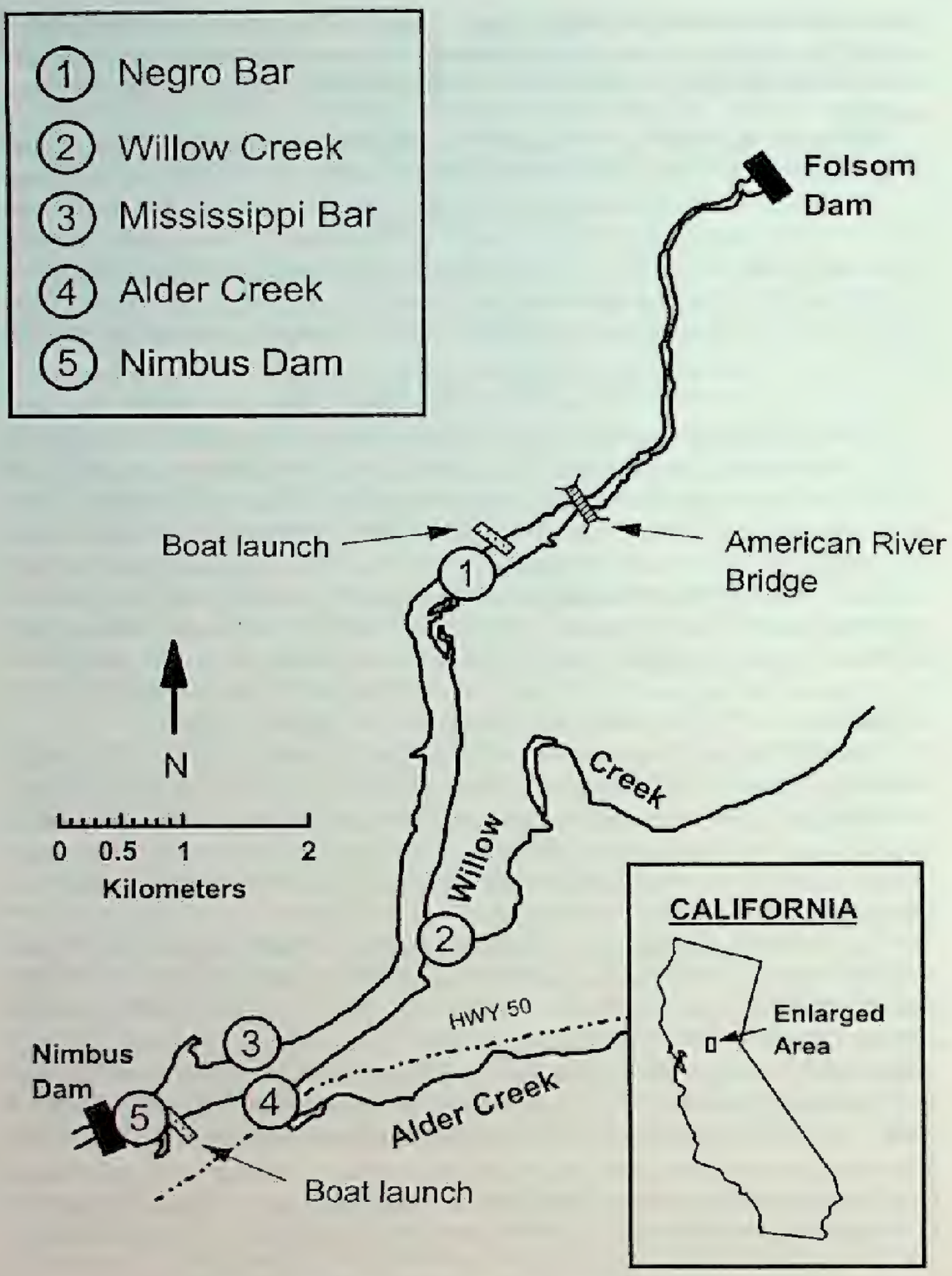

Figure 1. Map of the study area showing locations of sampling sites.

about 182 g each.

Immediately after capture, fish retained for mercury determinations were measured for total length (TL), then placed into clean plastic ziplock bags (along with unique code identifiers) and chilled on wet ice. Within 24 hrs, fish were weighed, then scale samples were taken for age determinations and stored in coin envelopes. Skinless fillets from both sides of each fish were obtained by dissection, then the two fillets were weighed and wrapped in clean plastic sheets, double-bagged in new plastic ziplock bags, and frozen (-10°C). The remaining carcass of each fish was then archived by wrapping and bagging in plastic before freezing.

Fish scales were analyzed to estimate the ages of only centrarchids (sunfish and bass) and rainbow trout. Multiple scales from each fish were soaked and cleaned in tapwater, dry-mounted between two glass microscope slides, then examined by viewing magnified images from a microprojector (Jearld 1983). Annuli were identified according to criteria described by Jearld (1983).

Total length and weight data were used to compute length-weight equations for each fish species by using the formula, $W=a*L^b$, where "a" is the intercept, "b" is the slope, W is fish weight, and L is fish length (Anderson and Neumann 1996). The length-weight equation was subsequently used to compute the relative condition factor (K_n) of individual fish (Anderson and Neumann 1996). Relative condition factor is computed from the formula, $K_n=W/W'$, where W is the actual weight and W' is estimated from the length-weight equation.

Frozen fish fillet samples where shipped overnight on dry ice to the USGS Columbia Environmental Research Center (CERC) in Columbia, Missouri, for moisture and total mercury determinations. Total mercury was measured because approximately 95-99% of the mercury in fish muscle tissues occurs as methylmercury (Grieb et al. 1990, Bloom 1992, Weiner and Spry 1996). Methods used to homogenize fillets varied according to biomass of the sample. Large fillets (>500 g) were processed through a Hobart band saw whereas intermediate-sized fillets (100-500 g) were homogenized with a Kitchen Aid blender and attached meat processor unit. Small fillets (20-100 g) were minced with a titanium knife. After homogenization, the samples were lyophilized with a Virtis Genesis 35EL freeze dryer for moisture determination. Once dried, samples weighing more than 3 g were further homogenized with a Bamix mixer/blender, whereas smaller samples were mechanically ground with a glass rod. Dried samples were stored in glass vials in a desiccator. Total mercury was determined with a direct mercury analyzer. A dried fish sample (50-100 mg) was first combusted in a stream of oxygen, then the volatilized sample was trapped by amalgamation on a gold substrate and thermally desorbed for quantification by atomic absorption spectrophotometry. The entire process was conducted with a Milestone DMA-80 analyzer equipped with an automated sample carousel.

For quality control purposes, the fish fillet samples were placed into 10 groups or blocks for mercury determinations. Quality control procedures included analysis of blanks, replicate samples, pre-combustion methylmercury spikes, and tissue reference materials. During instrumental runs, additional quality control procedures consisted of independent calibration verification checks. An independent calibration verification

standard (Spex Claritas PPT, Cat. No. CL1-49Hg, Spex CertiPrep, Metuchen, New Jersey) was analyzed at the beginning and end of each instrumental run to confirm the calibration status of the DMA-80 system. The percentage errors varied from –6.3 to 7.8. Results from analysis of three reference fish tissues—CERC wholebody striped bass, 2.29±0.05 µg Hg/g dry weight (henceforth, all references to mercury concentrations will refer to total mercury reported in terms of dry weight unless specified otherwise); National Institute of Standards and Technology Research Material (NIST) 50 albacore tuna fillet, 0.92±0.02 µg Hg/g; and National Research Council Canada (NRCC) DORM-2 CRM dogfish muscle, 4.77±0.09 µg Hg/g—were within certified or recommended means and ranges. Method precision, determined from analyses of 18 duplicate pairs of fish tissue samples, did not exceed 5.8% relative standard deviation (RSD); most were less than 3.0% RSD. Recovery of methylmercury from pre-combustion tissue spikes varied from 102% to 111% (mean, 108%). Blank equivalent concentrations were consistently less than the method detection limits (0.0007-0.0059 µg Hg/g), whereas the limits of quantitation were 0.0023-0.0177 µg Hg/g. The instrument detection limit was 0.066 ng Hg/ml. Within-run variability was 1% RSD. Overall, these quality control results were within acceptable limits as specified by CERC.

Data were analyzed with software such as SAS and SAS/GRAPH (SAS Institute Inc., Cary, North Carolina), and Lotus Freelance Graphics for Windows (Lotus Development Corp., Cambridge, Massachusetts). Many variables measured during this study were not normally distributed. Even when these variables were subjected to standard transformations (e.g., angular transformation for moisture content, logarithmic transformation for length, weight, K_n, age, and mercury concentration), normality of the data was not always achieved. Consequently, with two exceptions, nonparametric techniques (e.g., Spearman rank correlation, Kruskal-Wallis test) were used to describe relations between variables such as length or weight of fish, and moisture content and mercury concentrations in their fillets. The exceptions were computations of predictive equations for length-weight relationships and length-mercury relationships. When nonparametric techniques were not available for statistical comparisons, we followed the recommendation of Conover (1980) and used rank-transformed data with applicable parametric procedures (e.g., multiple means comparison tests, analysis of covariance). Unless specified otherwise, the level of significance for all statistical tests was P=0.05.

RESULTS

A total of 192 fish fillet samples was analyzed for moisture content and mercury concentrations during this study (Table 1; see Saiki et al.[6] [2004] for raw data from this and other recent surveys at Lake Natoma). Moisture content in various species ranged from 75.8% to 83.6%, whereas mercury concentrations ranged from 0.08 µg/g to 3.85

[6]Saiki, M.K., D.G. Slotton, T.W. May, S.M. Ayers, and C.N. Alpers. 2004. Summary of total mercury concentrations in fillets of selected sport fishes collected during 2000-2003 from Lake Natoma, Sacramento County, California. U.S. Geological Survey Data Series 103 (http://water.usgs.gov/pubs/ds/ds103/).

Table 1. Summary characteristics of fish species sampled from Lake Natoma during September-October 2002, and moisture content and total mercury concentrations in their fillets. Except for moisture, values are geometric means (ranges in parentheses). For moisture, values are back-transformed angular means (ranges in parentheses).

Fish species	N	Total length (mm)	Weight (g)	K_n	Age (years)	Moisture (%)	Mercury (µg/g wet weight)	Mercury (µg/g dry weight)
Green sunfish	4	118 (90-151)	30.8 (12.5-75.5)	1.00 (0.94-1.06)	1.4 (1-2)	81.4 (80.8-81.8)	0.11 (0.06-0.20)	0.58 (0.33-1.02)
Bluegill	97	116 (72-174)	25.2 (5.0-91.5)	1.00 (0.74-1.40)	1.5 (1-4)	81.8 (80.7-83.2)	0.08 (0.04-0.18)	0.45 (0.22-1.06)
Redear sunfish	20	127 (80-187)	33.7 (7.5-117.0)	1.00 (0.95-1.17)	1.5 (1-3)	80.8 (80.1-81.8)	0.07 (0.03-0.39)	0.40 (0.15-1.99)
Smallmouth bass	2	159 (145-174)	46.0 (35.0-60.5)	1.00 (1.00-1.00)	1.0 (1-1)	79.8 (79.0-80.6)	0.13 (0.11-0.16)	0.66 (0.51-0.84)
Spotted bass	3	228 (118-335)	141.9 (15.5-476.0)	1.00 (0.93-1.09)	1.6 (1-2)	79.3 (77.1-81.9)	0.27 (0.10-0.49)	1.29 (0.53-2.29)
Largemouth bass	61	212 (88-490)	126.3 (7.0-1,967.5)	1.01 (0.38-1.31)	1.8 (0-8)	79.7 (77.6-83.0)	0.21 (0.06-0.86)	1.02 (0.34-3.85)
White catfish	1	249	229.5	1.00	—	81.8	0.56	3.07
Black bullhead	1	214	134.0	1.00	—	83.6	0.15	0.88
Brown bullhead	1	317	554.0	1.00	—	82.1	0.35	1.97
Rainbow trout	2	239 (177-324)	145.6 (48.0-441.5)	1.00 (1.00-1.00)	0.4 (0-1)	78.9 (75.8-81.9)	0.04 (0.02-0.10)	0.21 (0.08-0.54)

µg/g. The species with highest concentrations of mercury included brown bullhead (1.97 µg/g), redear sunfish (1.99 µg/g), spotted bass (2.29 µg/g), white catfish (3.07 µg/g), and largemouth bass (3.85 µg/g).

Sufficient sample sizes were available for bluegill, redear sunfish, and largemouth bass to determine if moisture content and mercury concentrations in fillets varied with length, weight, K_n, and age, and if mercury concentrations varied spatially among sampling sites. Moisture content was inversely correlated with total length and weight in redear sunfish and largemouth bass, and with age in largemouth bass, whereas similar relationships were not observed in bluegill (Table 2). On the other hand, mercury concentrations were directly correlated with length, weight, and age in bluegill and largemouth bass, but not in redear sunfish. Moisture content and mercury concentrations were not significantly correlated with K_n values in the three fish species.

Table 2. Spearman rank correlation coefficients between total length, weight, relative condition factor (K_n), and age of bluegill, redear sunfish, and largemouth bass, and moisture content and total mercury concentrations in their fillets.[a]

Common name	N	Vital statistic	Moisture (%)	Mercury (µg/g dry weight)
Bluegill	97	Total length	-0.17	0.21*
		Weight	-0.18	0.22*
		K_n	<0.01	0.08
		Age	-0.16	0.47***
Redear sunfish	20	Total length	-0.63**	-0.18
		Weight	-0.60**	-0.21
		K_n	-0.36	0.12
		Age	-0.25	-0.06
Largemouth bass	61	Total length	-0.91***	0.90***
		Weight	-0.92***	0.91***
		K_n	-0.11	0.13
		Age	-0.61***	0.69***

[a]Codes: *, $P \leq 0.05$; **, $P \leq 0.01$; ***, $P \leq 0.001$.

To detect spatial variations in mercury content of bluegill and largemouth bass, mercury concentrations were adjusted for length, weight, and age of individual fish. No such adjustments were necessary for redear sunfish because length, weight, and age were not correlated with mercury in this species. Only bluegill exhibited significant spatial variations for mercury concentrations, with generally higher concentrations occurring in upstream sites (Negro Bar, Willow Creek, and Mississippi Bar) than in downstream sites (Alder Creek and Nimbus Dam) (Table 3). However, spatial variations were not apparent when mercury concentrations were adjusted for age. Redear sunfish and largemouth bass did not exhibit significant spatial variations in mercury concentrations (Tables 4-5).

Table 3. Total mercury concentrations in fillets of bluegill from various sites in Lake Natoma during September-October 2002. The mercury concentrations are least squares means (adjusted for total length, weight, or age of bluegill) computed by analysis of covariance (ANCOVA) with logarithmically transformed data. Also shown are ranges for mercury concentrations. Due to non-normal distribution of transformed data, F-statistics and means comparisons tests were generated with ranked data.[a,b,c]

Site or F-statistic	N	Mercury concentrations (µg/g dry weight)			
		Total length	Weight	Age	Range
Negro Bar	22	0.52 C	0.52 C	0.51 A	0.28-1.06
Willow Creek	9	0.47 AC	0.48 AC	0.48 A	0.22-0.90
Mississippi Bar	25	0.48 BC	0.48 BC	0.47 A	0.30-0.78
Alder Creek	19	0.39 A	0.39 A	0.41 A	0.26-0.60
Nimbus Dam	22	0.41 AB	0.41 AB	0.42 A	0.26-0.83
$F_{4,91}$		2.76*	2.69*	1.89	

[a]Before ANCOVA was conducted, all regression lines under comparison were determined to be parallel (for total length, $F_{4,87}$=0.89, P=0.4730; for weight, $F_{4,87}$=0.76, P=0.5563; for age, $F_{4,87}$=0.76, P=0.5546).
[b]Within a column, least squares means followed by the same capital letter are not significantly different (P>0.05).
[c]Code: *, P≤0.05.

Table 4. Total mercury concentrations in fillets of redear sunfish from various sites in Lake Natoma during September-October 2002. Geometric means (ranges in parentheses) were computed from logarithmically transformed data.

Site or Kruskal-Wallis statistic	N	Mean (range)
Negro Bar	5	0.47 (0.21-2.00)
Willow Creek	13	0.40 (0.16-0.89)
Mississippi Bar	2	0.28 (0.15-0.55)
χ^2, df=2		0.26 N.S.[a]

[a]N.S., not significant (P>0.05).

Table 5. Total mercury concentrations in fillets of largemouth bass from various sites in Lake Natoma during September-October 2002. The mercury concentrations are least squares means (adjusted for total length, weight, or age of bass) computed by analysis of covariance (ANCOVA) with logarithmically transformed data. Also shown are ranges for mercury concentrations. Due to non-normal distribution of transformed data, F-statistics were generated with ranked data.[a,b]

Site or F-statistic	N	Mercury concentrations (µg/g dry weight)			
		Total length	Weight	Age	Range
Negro Bar	16	0.98	0.98	1.09	0.50-3.24
Willow Creek	16	1.09	1.07	1.22	0.67-3.85
Mississippi Bar	6	1.00	1.03	0.75	0.39-3.66
Alder Creek	16	0.99	0.99	1.04	0.51-2.76
Nimbus Dam	7	1.01	1.03	0.87	0.34-1.10
$F_{4,55}$		1.11 N.S.	1.17 N.S.	1.81 N.S.	

[a]Before ANCOVA was conducted, all regression lines under comparison were determined to be parallel (total length, $F_{4,51}$=0.46, P=0.7679; weight, $F_{4,51}$=0.65, P=0.6290; age, $F_{4,51}$=0.74, P=0.5659).
[b]N.S, not significant (P>0.05).

DISCUSSION

A number of field investigations involving a variety of fish species indicate that mercury concentrations typically increase with size and age of fish (Lange et al. 1993, 1994; Wiener and Spry 1996). In Lake Natoma, the mercury concentrations in fillets of bluegill and largemouth bass, but not redear sunfish, also increased as fish increased in length, weight, and age (Table 2).

Although fish are exposed to mercury from both water and food, bioaccumulation of mercury through the food chain plays a more important role in determining fish mercury burdens and concentrations (Weiner and Spry 1996; Hall et al. 1997). Fish assimilate 15-20% of the methylmercury present in their forage (Phillips and Gregory 1979) and eliminate methylmercury slowly relative to the rate of uptake (Weiner and Spry 1996), resulting in a net increase in mercury burdens. MacCrimmon et al. (1983) observed increased rates of mercury accumulation in lake trout, *Salvelinus namaycush*, when the young switched from a diet of invertebrates to forage fish. Wren and MacCrimmon (1986) also observed that piscivorous fish had higher concentrations of mercury than prey fish of comparable age. In Lake Natoma, top-level predators such as largemouth bass, spotted bass, and white catfish had higher concentrations of mercury than did lower trophic level insectivores or planktivores such as bluegill, redear sunfish, and rainbow trout (Table 1). Moreover, smaller (younger) bass, which generally feed on zooplankton and small insects, contained lower concentrations of mercury than did larger (older) bass that feed primarily on large-bodied invertebrates (e.g., crayfish) and forage fish (Moyle 2002). Mercury concentrations were weakly

correlated with length, weight, and age of bluegill and not correlated with length, weight, and age of redear sunfish possibly because juveniles and adults of these species tend to forage on similar foods (e.g., zooplankton, immature aquatic insects, and other benthic invertebrates, although mollusks are usually less conspicuous in diets of bluegill than redear sunfish; Moyle 2002). Unlike adult largemouth bass, bluegill and redear sunfish are seldom piscivorous.

Body condition is often used as an easily measured index of growth, fatness, general "well-being," or gonad development (Anderson and Neumann 1996). Our study did not detect significant associations between relative condition factor (K_n) and mercury concentrations in fillets of bluegill, redear sunfish, and largemouth bass (Table 2). Nevertheless, Greenfield et al. (2001) hypothesized that an inverse relationship may exist between body condition and mercury concentration due to a phenomenon referred to as "growth dilution." According to Greenfield et al. (2001), fast growing fish reach a given size more efficiently, having spent proportionately less energy consuming and respiring to reach that size, and therefore have lower tissue contaminant concentrations per unit mass. Cizdziel et al. (2002) documented an inverse relation between body condition and mercury concentration in muscle plugs taken from mature striped bass, *Morone saxatilis*, in Lake Mead, a reservoir on the Colorado River. However, Cizdziel et al. (2002) speculated that the inverse relation was due to undernourished fish in poor body condition that lost muscle mass faster than the bound methylmercury, thus effectively increasing the mercury concentration in remaining tissue. By comparison, Friedman et al. (2002) reported no significant differences in body condition of largemouth bass from three New Jersey reservoirs even though the bass populations varied greatly in their body burdens of mercury.

Limited samples of mostly largemouth bass collected in August 2000 suggested that mercury concentrations in fish were not uniformly distributed throughout Lake Natoma (Bowman[2] 2002; also see Saiki et al.[6] [2004] for raw data). The evidence implied that higher concentrations may occur in fish from the mouth of Willow Creek than from the mouth of Alder Creek and other localities (Bowman[2] 2002). Although mean mercury concentrations in largemouth bass standardized for length, weight, and age were slightly higher in the mouth of Willow Creek than in four other sites during our study, the differences were not statistically significant (Table 5). By comparison, mean mercury concentrations standardized for length and weight of bluegill exhibited significant spatial variations within the lake, with higher mercury concentrations observed at upstream reservoir sites (Negro Bar, Willow Creek, and Mississippi Bar) than at downstream sites (Alder Creek and Nimbus Dam; Table 3). However, when mercury concentrations were standardized for age of bluegill, spatial variations were not significant. Redear sunfish did not exhibit spatial variations in samples collected from upstream sampling sites although none were collected downstream at Alder Creek and Nimbus Dam (Table 4). The relatively small surface area of Lake Natoma might allow sufficient mingling of some fish species from various portions of the reservoir to obscure easily discernible spatial patterns in mercury concentrations.

The U.S. Food and Drug Administration (USFDA) action level for methylmercury in fish is 1 μg/g wet weight, which is intended to limit exposure by humans to levels 10

times lower than the lowest levels associated with adverse health effects (USFDA[7] 1994). This action level is used to regulate the sale of commercially caught fish. In addition, USEPA recently established 0.3 mg methylmercury/kg wet weight as the fish tissue residue criterion that should not be exceeded to avoid undesirable neurological abnormalities in human infants exposed *in utero* when pregnant women consume mercury-contaminated foods (USEPA[3] 2001). None of the fish sampled from Lake Natoma during our study equaled or exceeded the USFDA action level (see mercury concentrations expressed as wet weight in Table 1). However, 1 of 20 redear sunfish, 14 of 61 largemouth bass, 1 of 1 brown bullhead, 2 of 3 spotted bass, and 1 of 1 white catfish exceeded the USEPA fish tissue residue criterion. Judging from our data, largemouth bass measuring 273 mm TL (about 292 g) would approximate the USEPA tissue residue criterion of 0.3 µg Hg/g wet weight (Fig. 2). Similar predictions for other fish species were not attempted due to small sample sizes or low R^2 values for regression models describing the relationship between length and mercury concentration in fillets.

In summary, results from this study indicated that total mercury concentrations in skinless fillets varied with length, weight, and age of largemouth bass and bluegill, but similar patterns were not observed in redear sunfish. In addition, mercury concentrations varied spatially in bluegill, with highest concentrations occurring in upstream reaches of Lake Natoma and lower concentrations occurring downstream. By comparison, spatial patterns were not apparent in redear sunfish and largemouth bass. Finally,

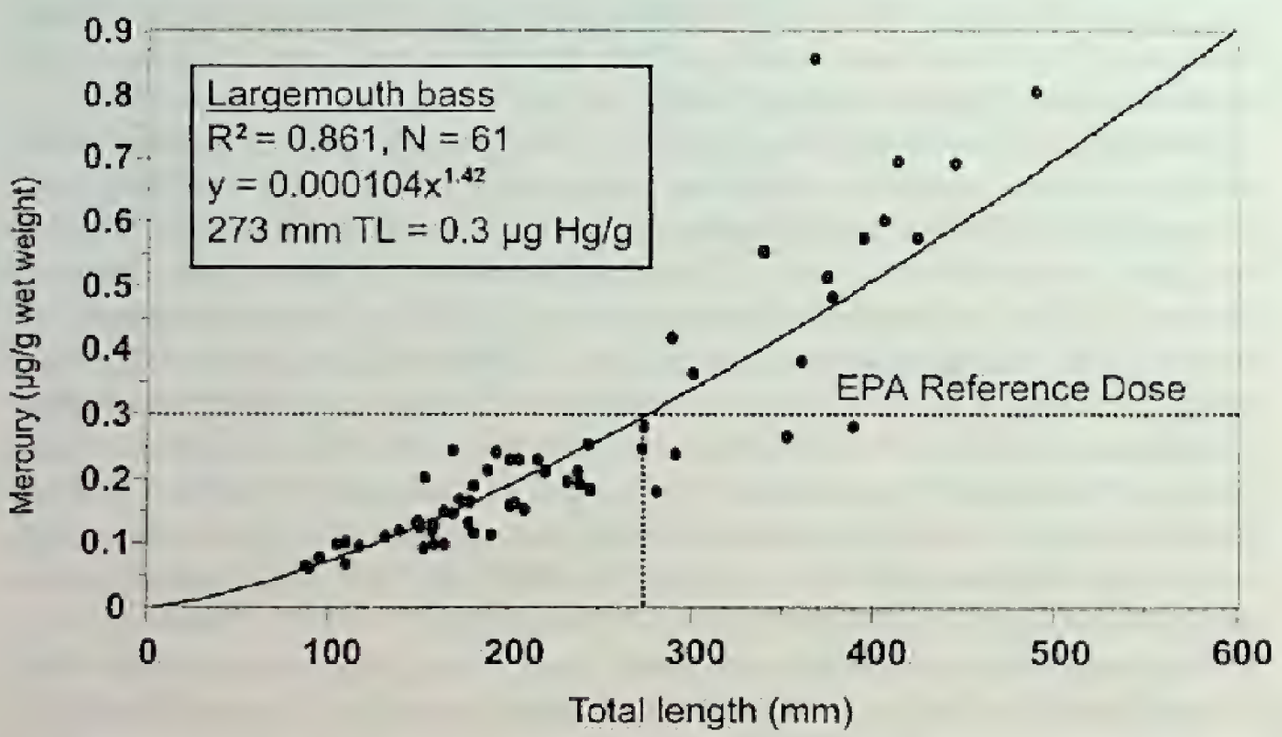

Figure 2. Relation between total length of largemouth bass and wet weight concentrations of total mercury in their skinless fillets.

[7]USFDA (U.S. Food and Drug Administration). 1994. Mercury in fish: cause for concern? FDA Consumer 28(7):unnumbered pages (http://www.fda.gov/fdac/reprints/mercury.html).

mercury concentrations in fillets did not equal or exceed 1.0 µg/g wet weight, the USFDA action level for methylmercury in seafood. However, mercury concentrations in fillets of some redear sunfish, largemouth bass, brown bullhead, and white catfish exceeded 0.3 µg/g wet weight, the USEPA tissue residue criterion for methylmercury. These results confirmed that some fish species inhabiting Lake Natoma are contaminated with undesirably high concentrations of mercury in their skinless fillets.

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LITERATURE CITED

Anderson, R.O., and R.M. Neumann. 1996. Chapter 15; length, weight, and associated structural indices. Pages 447-482 *in*: B.R. Murphy and D.W. Willis, editors. Fisheries techniques, second edition. American Fisheries Society, Bethesda, Maryland.

Bennett, G.W. 1971. Management of lakes and ponds, second edition. Van Nostrand Reinhold Company, New York.

Bloom, N.S. 1992. On the chemical form of mercury in edible fish and marine invertebrate tissue. Canadian Journal of Fisheries and Aquatic Sciences 49:1010-1017.

Bowie, A.J. 1905. A practical treatise on hydraulic mining in California. Van Nostrand, New York.

Cizdziel, J.V., T.A. Hinners, J.E. Pollard, E.M. Heithmar, and C.L. Cross. 2002. Mercury concentrations in fish from Lake Mead, USA, related to fish size, condition, trophic level, location, and consumption risk. Archives of Environmental Contamination and Toxicology 43:309-317.

Conover, W.J. 1980. Practical nonparametric statistics, second edition. John Wiley & Sons, New York.

Friedmann, A.S., E.K. Costain, D.L. MacLatchy, W. Stansley, and E.J. Washuta. 2002. Effect of mercury on general and reproductive health of largemouth bass (*Micropterus salmoides*) from three lakes in New Jersey. Ecotoxicology and Environmental Safety 52:117-122.

Greenfield, B.K., T.R. Hrabik, C.J. Harvey, and S.R. Carpenter. 2001. Predicting mercury levels in yellow perch: use of water chemistry, trophic ecology, and spatial traits. Canadian Journal of Fisheries and Aquatic Sciences 58:1419-1429.

Grieb, T.M., C.T. Driscoll, S.P. Gloss, C.L. Schofield, G.L. Bowie, and D.B. Porcella. 1990. Factors affecting mercury accumulation in fish in the upper Michigan peninsula. Environmental Toxicology and Chemistry 9:919-930.

Hall, B.D., R.A. Bodaly, R.J.P. Fudge, J.W.M. Rudd, and D.M. Rosenberg. 1997. Food as the dominant pathway of methylmercury uptake by fish. Water, Air, and Soil Pollution 100:13-24.

Jearld, A., Jr. 1983. Chapter 16: age determination. Pages 301-324 *in*: L.A. Nielsen and D.L. Johnson, editors. Fisheries techniques. American Fisheries Society, Bethesda, Maryland.

Lange, T.R., H.E. Royals, and L.L. Connor. 1993. Influence of water chemistry on mercury concentration in largemouth bass from Florida lakes. Transactions of the American Fisheries Society 122:74-84.

Lange, T.R., H.E. Royals, and L.L. Connor. 1994. Mercury accumulation in largemouth bass (*Micropterus salmoides*) in a Florida lake. Archives of Environmental Contamination and Toxicology 27:466-471.

Leitritz, E. 1970. A history of California's fish hatcheries: 1860-1970. California Department of Fish and Game, Fish Bulletin 150.

MacCrimmon, H.R., C.D. Wren, and B.L. Gots. 1983. Mercury uptake by lake trout, *Salvelinus namaycush*, relative to age, growth, and diet in Tadenac Lake with comparative data from other Precambrian shield lakes. Canadian Journal of Fisheries and Aquatic Sciences 40:114-120.

Moyle, P.B. 2002. Inland fishes of California, revised and expanded. University of California Press, Berkeley, California.

Phillips, G.R., and R.W. Gregory. 1979. Assimilation efficiency of dietary methylmercury by northern pike (*Esox lucius*). Journal of the Fisheries Research Board of Canada 36:1516-1519.

Wiener, J.G., and D.J. Spry. 1996. Toxicological significance of mercury in freshwater fish. Pages 297-339 *in*: W.N. Beyer, G.H. Heinz, and A.W. Redmon, editors. Environmental contaminants in wildlife—interpreting tissue concentrations. Lewis Publishers, Boca Raton, Florida.

Wren, C.D., and H.R. MacCrimmon. 1986. Comparative bioaccumulation of mercury in two adjacent freshwater ecosystems. Water Research 20:763-769.

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CHANGE IN AREA OF WINTER-FLOODED AND DRY RICE IN THE NORTHERN CENTRAL VALLEY OF CALIFORNIA DETERMINED BY SATELLITE IMAGERY

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ABSTRACT

To understand possible factors impacting distribution of wintering waterfowl, we analyzed satellite imagery to estimate area of winter-flooded (flooded and saturated soil) and winter-dry harvested rice in Butte, Colusa, American, Sutter, Yolo, and Delta basins in the northern Central Valley of California. We compared our results for the 1999-00 winter with estimates that Spell et al. (1995) reported using identical methods for 1988-89 and 1993-94. Area of winter-flooded rice in the northern Central Valley in 1999-00 (78,841 ha) was 37% greater than in 1993-94 (57,702 ha) and 46% greater than in 1988-89 (53,816 ha). Winter-flooded rice increased an average of 3,253 ha per year between 1993-94 and 1999-00 compared to 777 ha per year between 1988-89 and 1993-94. The increase in flooded rice area was due to both an increase in total rice area (201,512 ha in 1999-00 vs. 171,918 ha in 1993-94 and 163,586 ha in 1988-89) and an increase in percentage of rice area that was flooded (39% in 1999-00 vs. 34% in 1993-94 and 33% in 1988-89). Change in winter-flooded rice area varied among basins. These results will help understand changes in distribution of wintering waterfowl among Central Valley basins during 1988-2000. The challenge for wildlife managers is to develop strategies that incorporate impacts of the increase in rice area in the northern Central Valley while recognizing that various factors could reduce rice area in the future.

INTRODUCTION

The Central Valley of California is a critical wintering area for many species of waterfowl and other wetland-dependent birds in the Pacific Flyway (Gilmer et al. 1982,

Heitmeyer et al. 1989). Once estimated at 1.6 - 2 million hectares, Central Valley wetlands were reduced by over 90% by the early 1900s (United States Fish and Wildlife Service 1978[1]). In the northern Central Valley, rice fields replaced many wetlands, and when flooded, maintain many of the same functions as wetlands they replaced (Elphick 2000). Both winter-flooded and winter-dry harvested rice fields are used heavily by waterfowl and other birds but species richness and use by ducks is greater in winter-flooded rice fields (Day and Colwell 1998, Elphick and Oring 1998).

Conservation programs such as the Central Valley Joint Venture that began in 1986 (Central Valley Habitat Joint Venture Implementation Board 1990[2]), the California Rice Straw Burning Reduction Act of 1991 (Assembly Bill 1378) that mandated the gradual phase-out of rice straw burning (Hill et al. 1999[3]), and changing agricultural practices and programs have impacted the amount of planted and winter-flooded rice. Tracking these changes is necessary to understand their impacts on ecology of wildlife that use rice fields. For example, information on change in area and distribution of winter-flooded rice fields is needed to understand factors related to wintering waterfowl distribution and aid the Central Valley Joint Venture in meeting its goal of providing habitat adequate to support and maintain historic distribution of wintering waterfowl.

To determine change in rice field habitats during 1988-2000, we analyzed late-summer satellite imagery to determine rice field area and mid-winter imagery to estimate area of winter-flooded (flooded and saturated soil) and winter-dry rice in Butte, Colusa, American, Sutter, Yolo, and Delta basins in the northern Central Valley of California during the 1999-00 season. We then compared our results with estimates that Spell et al. (1995) reported using identical methods for the 1988-89 and 1993-94 seasons.

STUDY AREA

The Central Valley of California (including the Suisun Marsh) is composed of nine basins (Fig. 1, United States Fish and Wildlife Service 1978[1]), but about 95% of the rice area is in the northern six basins (Butte, Colusa, American, Sutter, Yolo, and Delta) (United States Department of Agriculture 2004[4]) where we determined rice area present and flooded during mid-winter.

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[1]United States Fish and Wildlife Service. 1978. Concept plan for waterfowl wintering habitat preservation, Central Valley, California. United States Fish and Wildlife Service, Portland, Oregon, USA.

[2]Central Valley Habitat Joint Venture Implementation Board. 1990. Central Valley Habitat Joint Venture Implementation Plan, North American Waterfowl Management Plan, Portland, Oregon, USA.

[3]Hill, J. E., D. M. Brandon, and S. M. Brouder. 1999. Agronomic implications of alternative rice straw management practices. Agronomy Progress Report Number 264. University of California, Davis, California, USA.

[4]United States Department of Agriculture. 2004. NASS Agricultural Statistics Data Base: Crops county and district data. (http://www.nass.usda.gov:81/ipedbcnty/report2.htm).

Figure 1. Basins in the north Central Valley where rice area was mapped from a 23 July 1999 satellite image and winter-flooded rice was mapped from a 30 December 1999 image.

Water availability and precipitation differed during 1988-89, 1993-94, and 1999-00. October – September water availability for the northern Central Valley, which varies with water supplies remaining from the previous year and current year precipitation, was "critical" (i.e., critically dry) in 1988-89, "above average" in 1993-94, and "wet" in 1999-00 (Gehrts 2002[5]). October-September rainfall at Sacramento Executive Airport was 42.6 cm during 1988-89, 30.8 cm during 1993-94, and 55.4 cm during 1999-00 (National Oceanic and Atmospheric Administration 2004[6]). However, precipitation was minimal before winter satellite imagery was collected each year with slightly less rain falling at Sacramento Executive Airport 0-3 days, 0-7 days, 0-14 days, and 0-30 days before 30 December 1999 (0.0, 0.0, 0.0, 0.15 cm, respectively) than before 24 January 1989 (0.33, 0.33, 0.33, 2.90 cm, respectively), and 6 January 1994 (0.15, 0.15, 0.28, 4.64 cm, respectively).

METHODS

We replicated methodology used by Spell et al. (1995) to process satellite imagery and estimate area of flooded (flooded and saturated soil) and non-flooded rice in Butte, Colusa, American, Sutter, Yolo, and Delta basins. Like Spell et al. (1995), in addition to flooded areas, we included areas of saturated soil in our winter-flooded estimate (<10% of total) to account for fields just beginning to flood or just after water was drained. The lack of rainfall for 2 weeks before winter imagery was collected eliminated the likelihood of misclassifying rain-wetted soils as soils that were saturated due to flooding. Our imagery covered the entire area of the six northern Central Valley basins whereas imagery used by Spell et al. (1995) did not include the northern 25% of Colusa (37 ha of rice present during our study) and Butte (300 ha of rice present during our study) basins, and the southern 48% of the Delta basin (1,292 ha of rice present during our study). Like Spell et al. (1995), we classified Landsat Thematic scenes acquired in late summer (23 July 1999) into rice and non-rice classes and scenes acquired in mid-winter (30 December 1999) to identify winter-flooded (flooded and saturated soil) and winter-dry areas.

We confirmed classification of areas using data provided by managers, aerial photographs, and site visits. We used a Geographic Information System (GIS) and ARC/GIS (ESRI, Redlands, California, USA) computer program to overlay a digitized basin coverage provided by the Central Valley Joint Venture on the classified imagery and derive basin estimates. We compared our 1999-00 results with estimates that Spell et al. (1995) reported for 1988(-89) using 1 August 1988 and 24 January 1989 imagery and for 1993(-94) using 30 July 1993 and 6 January 1994 imagery.

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[5]Gehrts, K. 2002. Water year hydrologic classification indices for the Sacramento and San Joaquin Valleys. IEP Newsletter: Interagency Ecological Program for the San Francisco Estuary 15:16-17. (http://www/iep.water.ca.gov/report/newsletter/2002winter/IEPNewsletterwinter2002.pdf).

[6]National Oceanic and Atmospheric Administration. 2004. Sacramento Executive Airport: Hourly/Daily Data, Local Climatological Data. (http://ols.ncdc.noaa.gov/cgi-bin/nndc/genOL-002.cgi).

RESULTS

Area of winter-flooded rice in the northern Central Valley in 1999-00 (78,841 ha) was 37% greater than in 1993-94 (57,702 ha) and 46% greater than in 1988-89 (53,816 ha) (Table 1). The increase in flooded rice area was due to both an increase in total rice area (201,512 ha in 1999-00 vs. 171,918 ha in 1993-94 and 163,586 ha in 1988-89) and an increase in percentage of rice area that was flooded (39% in 1999-00 vs. 34% in 1993-94 and 33% in 1988-89). The average annual increase in winter-flooded rice area in the northern Central Valley between 1993-94 and 1999-00 was 3,523 ha compared to 777 ha between 1988-89 and 1993-94.

Change in winter-flooded rice area varied among basins (Table 1). Between 1993-94 and 1998-00, winter-flooded rice area increased in Butte (+24%), Colusa (+78%), American (+28%), and Sutter (+22%) basins due to an increase in both rice area and the percent flooded but decreased in Yolo (-6%) due to an increase in rice area but decrease in percent flooded. In the Delta basin, rice area declined but percent flooded increased, resulting in little change (-2%) in winter-flooded rice area. Between 1988-89 and 1993-94, winter-flooded rice area increased in Butte (+30%) and Colusa (+16%) basins due to an increase in both rice area and the percent flooded but decreased in American (-11%), Sutter (-15%), Yolo (-33%), and Delta (-82%) basins because rice area (in Yolo), percent flooded (in American), or both (in Sutter and Delta) declined.

Table 1. Total area of rice and area and percentage that was winter-flooded in each northern Central Valley basin during 1999-00 vs. 1993-94, and 1988-89[a].

	1999-00			1993-94			1988-89		
Basin	Total ha	Flooded ha	Flooded %	Total ha	Flooded ha	Flooded %	Total ha	Flooded ha	Flooded %
Butte	54,669	30,659	56	47,408	24,718	52	45,399	18,977	42
Colusa	80,528	24,381	30	65,752	13,674	21	61,858	11,742	19
American	39,232	15,564	40	34,764	12,158	35	27,942	13,701	49
Sutter	20,956	6,624	32	18,374	5,444	30	18,442	6,399	35
Yolo	4,694	1,497	32	3,425	1,590	46	5,206	2,359	45
Delta	1,433	116	8	2,195	118	5	4,739	638	13
All	201,512	78,841	39	171,918	57,702	34	163,586	53,816	33

[a]Winter-flood values for 30 December 1999, 6 January 1994, or 24 January 1989. Values for 1999-00 from this study, values for 1988-89 and 1993-94 from Spell et al. (1995).

DISCUSSION

Our data indicate that both winter-flooded and total rice area in the northern Central Valley was greater in 1999-00 than in 1993-94, continuing an increase that occurred between 1988-89 and 1993-94. However, the increase in the area of winter-flooded rice

during the late 1990s was greater than during the previous five years because both the percentage of rice that was winter-flooded and the total area of rice increased. No rain fell during the two weeks prior to collection of the 1999-00 winter satellite image and rainfall was minimal before all the winter images. Thus, the greater percentage and area of winter-flooded rice in 1999-00 than earlier years was clearly due to an increase in flooding from delivered (i.e., managed) water rather than rainfall.

Improved rice farming economics due to improved rice prices and subsidies was probably the main factor for increased rice area (Childs 1997). We speculate that a number of factors resulted in a greater proportion and area of winter-flooded rice in the Sacramento Valley basins during 1999-00 than earlier years, including: (1) increasing restrictions on rice straw burning that started in 1992 (Hill et al. 1999[3]); (2) conservation payments for infrastructure allowing rice straw rolling and flooding that started in 1997 (Ducks Unlimited, Inc., 2004[7]); (3) new information showing utility of flooding and waterfowl presence for straw decomposition (Wrysinski et al. 1995[8], Bird et al. 2000); (4) increased opportunity for hunting revenue on flooded fields (Passaglia 1997[9]) due to greater hunter demand for leased blinds with improved waterfowl populations (e.g., 3.99 million waterfowl in 1993 vs. 6.95 million waterfowl in 1999 during Pacific Flyway midwinter survey; Trost and Drut 2003[10]) and more liberal harvest regulations (e.g., 59-day season and 5-duck daily bag limit in 1989 and 1993 vs. 100-day season and 7-duck daily bag limit in 1999; California Department of Fish and Game 1989[11], 1993[12], 1999[13]); and (5) better winter water availability (California Department of Water Resources 1991[14], Davis 1992, Gehrts 2002[5]).

Although Spell et al. (1995) did not study the northern 25% of the Colusa and Butte basins, while we included the entire area of both these basins, the 337 ha of rice we found

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[7]Ducks Unlimited, Inc. 2004. Valley Bay CARE: CVPIA (b)(22) Private Land Winter Flooding. (http://www.ducks.org/conservation/valleycare.asp).

[8]Wrysinski, J. E., J. D. Garr, and M. A. Bias. 1995. Valley Habitats Number 1: Rice straw decomposition and development of seasonal waterbird habitat on rice fields. Ducks Unlimited Western Regional Office, Sacramento, California, USA.

[9]Passaglia, M. A. 1997. Ways to lease your land. Pages 3-4 *in* Anonymous, editor, Valley Habitats Number 16: Establishing a hunting program on your farm. Ducks Unlimited Western Regional Office, Sacramento, California, USA.

[10]Trost, R. E., and M. S. Drut. 2003. 2003 Pacific Flyway Data Book. United States Fish and Wildlife Service, Portland, Oregon, USA.

[11]California Department of Fish and Game. 1989. 1989 California hunting regulations: Parts II and III. Resident and migratory game birds. California Department of Fish and Game, Sacramento, California, USA.

[12]California Department of Fish and Game. 1993. 1993 California hunting regulations: Parts II and III. Resident and migratory game birds. California Department of Fish and Game, Sacramento, California, USA.

[13]California Department of Fish and Game. 1999. 1999 California hunting regulations: Parts II and III. Resident and migratory game birds. California Department of Fish and Game, Sacramento, California, USA.

[14]California Department of Water Resources. 1991. California's continuing drought, 1987-1991: A summary of impacts and conditions as of December 1, 1991. Sacramento, California, USA.

in this additional area represented only 0.2% (337 ha/135,197 ha) of the total rice area in these basins and accounted for only 1.5% (337 ha/[135,197 ha – 113,160 ha]) of the increase in total and flooded rice area between 1993-94 vs. 1999-00 in these basins. The 5.1% increase between 1988-89 and 1993-94 reported by Spell et al. (1995) and the 17.2% increase we found between 1993-94 and 1999-00 in total rice area in northern Central Valley basins based upon classification of satellite imagery is consistent with the increase in planted rice area reported for the Central Valley during those periods (2.5% increase between 1988-89 and 1993-94 and 15.9% increase between 1993-94 and 1999-00, United States Department of Agriculture 2004[4]).

The reason(s) that the area of rice grown in the Delta basin apparently decreased while rice area in most other northern Central Valley basins increased is unknown. Most rice we mapped in the Delta basin during 1999-00 was outside Spell et al.'s (1995) coverage, indicating that the decline of rice between 1993-94 and 1999-00 inside the Delta area covered by Spell et al. (1995) was even greater than what our estimate for the entire Delta basin shows. Perhaps the cooler Delta climate, along with other regional differences (e.g., soils), resulted in economics of rice being less favorable than other crops and conversion of rice fields to other agriculture (e.g., orchards, vineyards). Alternatively, the cooler climate and different rice varieties grown in the Delta (California Rice Research Board 2001[15]) may have resulted in different timing of rice planting, flooding, and growth, leading to us misclassifying some Delta rice as other crops or Spell et al. (1995) misclassifying some Delta non-rice crops as rice fields. Rice area planted in San Joaquin County, which encompasses most of the Delta basin and no other basin, reportedly ranged between about 1400 ha to 2000 ha each year during 1982-2000, with no obvious declining trend (United States Department of Agriculture 2004[4]).

We caution that our and Spell et al.'s (1995) estimates of total and winter-flooded rice area are single year estimates, and although total and winter-flooded rice increased between 1988-89, 1993-94, and 1999-00, changing economics, water availability, increased mosquito abatement costs (Lee 2004[16]), and other factors (e.g., water sales to southern California; Helton 2003) could quickly reverse this apparent trend. Further, each of our winter flooding estimates is based upon a single, mid-winter satellite image, and while representative of conditions during mid-winter, overlooks any differences in flooding occurring during early- and late-winter.

MANAGEMENT IMPLICATIONS

Waterfowl, like many other avian species, are highly mobile and can quickly respond to changes in distribution of preferred habitats (Heitmeyer et al. 1989, Fleskes

[15]California Rice Research Board. 2001. Annual Report: Rice Breeding Program-01. (http://www.syix.com/rrb/01rpt/Breeding.htm).

[16]Lee, M. 2004. The mosquito zone: Pressure on rice growers to help pay for fight against West Nile virus. Sacramento Bee, Business Section, Friday, July 23. (http://www.sacbee.com/content/business/story/10100327p-11021164c.html).

et al. 2002). Understanding how area of winter-flooded and winter-dry rice in the Central Valley, which are important habitats for waterfowl and many other birds (Miller 1987, Day and Colwell 1998, Elphick and Oring 1998, Elphick 2004), has changed during the 1988-2000 period will be useful for understanding changes in distribution of wintering waterfowl and other species during that interval. Historically, about 60% of the estimated 1.6 - 2 million hectares of wetlands originally in the Central Valley (United States Fish and Wildlife Service 1978[1], Gilmer et al. 1982) occurred in the Suisun, Delta, and other northern Central Valley basins (Colusa, Butte, American, Sutter, and Yolo), with the remaining 40% occurring in the San Joaquin and Tulare basins comprising the San Joaquin Valley. The conversion of wetlands in the northern Central Valley to rice fields, rather than to agriculture of less waterfowl value like occurred in the San Joaquin Valley, reduced the negative impacts of wetland loss in the northern Central Valley. Although nearly all winter-flooded rice is hunted for waterfowl (J. D. Garr, Ducks Unlimited, Inc., Sacramento, California, personal communication), thereby reducing its use as daytime waterfowl sanctuary to the degree it is hunted, all is fully available for nocturnal feeding. Thus, the increase in managed winter-flooding of rice that we report between 1988-89 and 1999-00 has increased the amount of feeding and loafing habitat available in the northern Central Valley landscape for most waterfowl and other wetland-dependent birds. Elphick (2004) reported that while winter flooding of harvested rice fields potentially has negative impacts for raptors and some other landbirds, it had no measurable impact on most species studied and possible benefit for some passerines. The increase in dry rice fields has increased the amount of foraging habitat available to some granivorous landbirds (Elphick 2004) and waterfowl such as geese that commonly use both flooded and dry rice fields (Elphick and Oring 1998). Given that birds are highly mobile and can respond to habitat changes over a wide range, we predict that unless area of wetland and agricultural habitats preferred by waterfowl and other birds increases at a similar scale in the San Joaquin Valley, the percentage of Central Valley waterfowl and birds that use agricultural fields will also increase in the northern Central Valley basins. The challenge for wildlife resource managers in the Central Valley is to develop management strategies that incorporate the current wildlife impact of the increase in total and winter-flooded rice area in the northern Central Valley while recognizing that changes in farm economics or other factors beyond their control could relatively quickly reduce area of planted and/or winter-flooded rice.

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LITERATURE CITED

Bird, J. A., G. S. Pettygrove, and J. M. Eadie. 2000. The effects of waterfowl foraging on rice straw decomposition: mutual benefits for waterfowl and growers. Journal of Applied Ecology 37:728-741.

Childs, N. 1997. High prices pull up U.S. rice area. Agricultural Outlook (September): 10-12. (http://www.ers.usda.gov/publications/agoutlook/sep1997/ao244.pdf).Davis, P. A. 1992. Omnibus western water law. Congressional Quarterly. November 21, 3687-3690.

Day, J. H., and M. A. Colwell. 1998. Waterbird communities in rice fields subjected to different post-harvest treatments. Colonial Waterbirds 21:185-197.

Elphick, C. S. 2000. Functional equivalency between rice fields and seminatural wetland habitats. Conservation Biology 14:181-191.

Elphick, C. S. 2004. Assessing conservation trade-offs: Identifying the effects of flooding rice fields for waterbirds on non-target bird species. Biological Conservation 117:105-110.

Elphick, C. S., and L. W. Oring. 1998. Winter management of California rice fields for waterbirds. Journal of Applied Ecology 35:95-108.

Fleskes, J. P., R. L. Jarvis, and D. S. Gilmer. 2002. Distribution and movements of female northern pintails radio tagged in the San Joaquin Valley, California. Journal of Wildlife Management 66:138-152.

Gilmer, D. S., M. R. Miller, R. D. Bauer, and J. R. LeDonne. 1982. California's Central Valley wintering waterfowl: concerns and challenges. Transactions of the North American Wildlife and Natural Resources Conference 47:441-452.

Heitmeyer, M. E., D. P. Connelly, and R. L. Pederson. 1989. The Central, Imperial, and Coachella valleys of California. Pages 475-505 *in* L. M. Smith, R.L. Pedersen, and R.M. Kaminski, editors. Habitat management for migrating and wintering waterfowl in North America. Texas Tech Press, Lubbock, Texas, USA.

Helton, T. L. 2003. Production: 2002 review. Rice Journal 106:1-4. (http://www.ricejournal.com/backissues/january2003/story1.asp)

Miller, M. R. 1987. Fall and winter foods of northern pintails in Sacramento Valley, California. Journal of Wildlife Management 51:405-414.

Spell, R., R. Kempka, and F. Reid. 1995. Evaluation of winter flooding of ricelands in the Central Valley of California using satellite imagery. Pages 357-366 *in* K. L. Campbell, editor. Versatility of wetlands in the agricultural landscape. American Society of Agricultural Engineers.

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CORRECT NOMENCLATURE FOR SIERRA NEVADA BIGHORN SHEEP

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Based on four specimens, Grinnell (1912) recognized bighorn sheep from the Sierra Nevada of California as a distinct subspecies, *Ovis cervina (= canadensis) sierrae*, and designated a 5 year-old male as the type specimen. In his revision of the taxonomy of North American sheep, Cowan (1940:556) did not recognize *sierrae* as a valid subspecies. While he acknowledged uncertainties about the taxonomic position of bighorn sheep inhabiting the Sierra Nevada, he listed and mapped them as belonging to *O. c. californiana*. Subsequent authors (Jones 1950, Hall and Kelson 1959, Manville 1980, Shackleton 1985) similarly have included bighorn sheep from the Sierra Nevada as *O. c. californiana*.

Recent analyses (Wehausen and Ramey 1993, 2000) have resulted in changes to Cowan's (1940) taxonomy. On the basis of genetic and morphometric data, Wehausen and Ramey (2000) reassigned specimens from north of the central Sierra Nevada, formerly designated as *O. c. californiana*, to *O. c. nelsoni* and *O. c. canadensis*, but retained the name *O. c. californiana* for bighorn sheep in the southern and central Sierra Nevada. That action raised the question of the correct subspecific name for bighorn sheep occupying the Sierra Nevada.

Douglas (1829) first applied the name California to bighorn sheep when he defined *Ovis californianus* based on a specimen from near Mount Adams, Washington that he obtained from natives, and designated the type specimen for this taxon (Shackleton 1985). His discussion of its range was, "... it appears to inhabit the subalpine regions of Mounts Wood [= Hood], St. Helens, and Vancouver, but is more numerous in the mountainous districts in the interior of California." How far south he considered this animal to range is not clear, and subsequent authors (Blyth 1840, Biddulph 1885, Bailey

1936) questioned the validity of his description and associated information. Douglas actually saw no sheep in the area he visited and based his description in part on a skin possessed by a native (Douglas 1829). Among the measurements he provided was a tail length of 18 inches, which Blyth (1840) and Biddulph (1885) noted to greatly exceed the tail length of any bighorn sheep. As a result, Biddulph (1885) questioned the legitimacy of *O. californianus*. Similarly, Bailey (1936) criticized the use of second-hand information, and questioned whether native sheep ever occupied the specific mountains in Washington that Douglas (1829) included in his description, given the high snowfall typical of those areas.

Although some early authors (Allen 1912, Lydekker 1913) recognized the subspecies *O. c. californiana*, Cowan (1940) was the first to assign specimens south of northeastern California to that subspecies. With the reassignment of specimens from north of the central Sierra Nevada to *O. c. nelsoni* and *O. c. canadensis* (Wehausen and Ramey 2000), Douglas' (1829) type specimen of *O. c. californiana* from Washington is no longer valid. By the Principle of Typification (International Commission on Zoological Nomenclature 1999), the type specimen designated by Grinnell (1912) replaces it, and the correct name for native sheep in the southern and central Sierra Nevada is *O. c. sierrae* Grinnell. We suggest the most appropriate common name for this subspecies to be Sierra Nevada bighorn sheep.

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LITERATURE CITED

Allen, J. A. 1912. Historical and nomenclatural notes on North American sheep. American Museum of Natural History Bulletin 31:1-29.

Bailey, V. 1936. The mammals and life zones of Oregon. United States Department of Agriculture, Bureau of Biological Survey, North American Fauna 55:1-416.

Biddulph, J. 1885. On the geographic races of the Rocky-Mountain bighorn. Proceedings of the Zoological Society of London 1885:678-684.

Blyth, Mr. 1840. An amended list of the species of the genus *Ovis*. Proceedings of the Zoological Society of London 86:62-81.

Cowan, I. McT. 1940. Distribution and variation in the native sheep of North America. American Midland Naturalist 24:505-580.

Douglas, D. 1829. Observations on two undescribed species of North American mammalia. Zoological Journal 4:332.

Grinnell, J. 1912. The bighorn of the Sierra Nevada. University of California, Publications in Zoology 10:143-153.

Hall, E. R., and K. R. Kelson. 1959. The mammals of North America. Ronald Press, New York, USA.

International Commission on Zoological Nomenclature. 1999. International code of zoological nomenclature, fourth edition. International Trust for Zoological Nomenclature, London, United Kingdom.

Jones, F. L. 1950. A survey of the Sierra Nevada bighorn. Sierra Club Bulletin 35(6):29-76.

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